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Potential Health and Environmental Hazards of Uranium Mine Wastes. Volume 2

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This report was prepared in response to Section 114(c) of Public Law 95-604 dated November 8, 1978 (USC78). This Section of the Law stipulates that, ´Not later than January 1, 1980, the Administrator, in consultation with the Commission, shall provide to the Congress a report which identifies the location and potential health, safety, and environmental hazards of uranium mine wastes together with recommendations, if any, for a program to eliminate these hazards.´ The purpose of this report is to comply fully with this request, as accurately and completely as available information will permit.

KEYWORDS: *Environmental impacts, *Uranium mines, *Radioactive wastes, *Health hazards.

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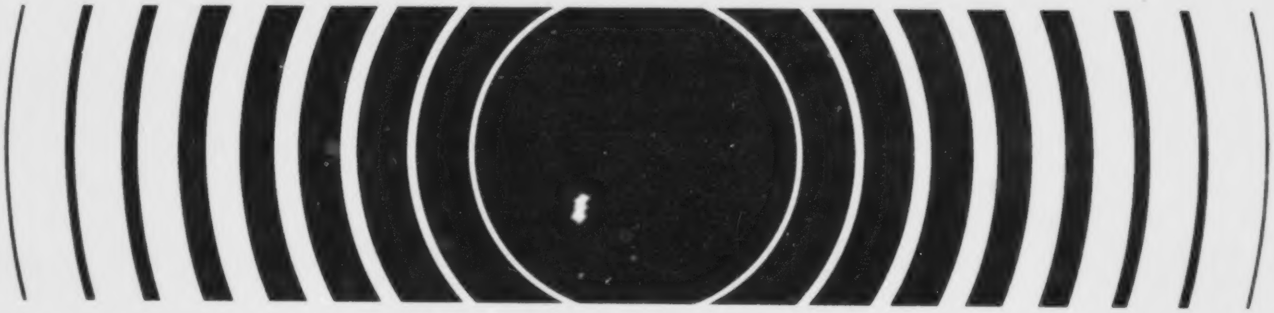
Potential Health and Environmental Hazards of Uranium Mine Wastes

Report To The Congress
Of The United States



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POTENTIAL HEALTH AND ENVIRONMENTAL HAZARDS OF URANIUM MINE WASTES

A Report to the Congress of the United States
in Response to Public Law 95-604

June 10, 1983

U.S Environmental Protection Agency
Office of Radiation Programs
Washington, D.C. 20460

CONTENTS

Page

Foreword xxix

1.0 Introduction 1-1

1.1 Purpose 1-1

1.1.1 Contents 1-1

1.2 Uranium Ore Production and Future Uranium Needs . . 1-2

1.2.1 Past Production 1-2

1.2.2 Projected Needs for Uranium 1-2

1.3 Overview of Uranium Mining Operations 1-5

1.3.1 General 1-5

1.3.2 Surface Mining 1-12

1.3.3 Underground Mining 1-16

1.3.4 In Situ Leaching 1-19

1.3.5 Other Mining Methods 1-27

1.3.5.1 Heap Leaching 1-27

1.3.5.2 Mine Water Recirculation 1-28

1.3.5.3 Borehole Slurry Mining 1-29

1.3.5.4 Uranium as a By-Product 1-30

1.4 Current Applicable Standards and Regulations . . 1-30

1.4.1 Federal Regulations 1-30

1.4.1.1 Federal Laws, Regulations and Guides for Protection of Health and Environment . . 1-31

1.4.1.1.1 Air Quality 1-31

1.4.1.1.2 Water Quality 1-33

1.4.1.1.3 Land Quality 1-34

1.4.1.2 Federal Mineral Leasing and Location/Patent Laws 1-35

1.4.1.2.1 Prospecting and Mining Rights . . 1-36

1.4.1.2.2 Mining and Environmental Plans . . 1-36

1.4.1.3 Laws Having Potential Applicability . . 1-36

Page

1.4.2 State Regulations 1-42
1.4.2.1 Colorado 1-45
1.4.2.2 New Mexico 1-47
1.4.2.3 Texas 1-49
1.4.2.4 Utah 1-49
1.4.2.5 Washington 1-50
1.4.2.6 Wyoming 1-51
1.5 References 1-53
2.0 Inventory of Uranium Mines 2-1
2.1 References 2-16
3.0 Potential Sources of Contaminants to the Environment and Man 3-1
3.1 Background Concentrations of Radionuclides and Trace Metals 3-1
3.1.1 Naturally Occurring Radionuclides 3-1
3.1.2 Stable Elements 3-6
3.2 Water-Related Aspects of Uranium Mining 3-11
3.2.1 Previous and Ongoing Hydrologic and Water Quality Studies Related to Uranium Mining 3-11
3.2.2 Mine Water Management 3-13
3.2.3 Water Quality Effects of Mine Water Discharge . 3-23
3.2.3.1 Behavior of Contaminants in the Aqueous Environment 3-23
3.2.3.1.1 Dilution and Suspended Sediment Transport 3-25
3.2.3.1.2 Sorption and Desorption 3-25
3.2.3.1.3 Precipitation 3-28
3.2.3.1.4 Biological Assimilation and Degradation 3-30
3.2.3.1.5 Complexation 3-31
3.2.3.2 Results of Field Studies in Uranium Mining Areas 3-32

Page

3.2.3.2.1 Colorado 3-32
3.2.3.2.2 Wyoming 3-34
3.2.3.2.3 Texas 3-38
3.2.3.2.4 New Mexico 3-39
3.2.3.3 Summary 3-43
3.3 Surface Mining 3-44
3.3.1 Solid Wastes 3-44
3.3.1.1 Overburden Piles 3-45
3.3.1.2 Ore Stockpiles 3-54
3.3.1.3 Sub-Ore Piles 3-60
3.3.1.4 Reclamation of Overburden Piles 3-62
3.3.2 Mine Water Discharge 3-63
3.3.2.1 Data Sources 3-63
3.3.2.2 Quantity and Quality of Discharge . . . 3-64
3.3.3 Hydraulic and Water Quality Effects of Surface Mine Discharge 3-68
3.3.3.1 Runoff and Flooding in the Model Surface Mine Area 3-68
3.3.3.1.1 Study Approach 3-68
3.3.3.1.2 Description of Area 3-70
3.3.3.1.3 Method of Study 3-72
3.3.3.1.4 Discussion of Results 3-77
3.3.3.2 Impacts of Seepage on Groundwater 3-89
3.3.4 Gases and Dusts from Mining Activities . . . 3-93
3.3.4.1 Dusts and Fumes 3-93
3.3.4.2 Radon-222 from the Pit, Storage Piles, and Ore Handling 3-99
3.4 Underground Mining 3-107
3.4.1 Solid Wastes 3-107
3.4.1.1 Waste Rock Piles 3-109
3.4.1.2 Ore Stockpiles 3-110
3.4.1.3 Sub-Ore Piles 3-112

Page

3.4.2 Mine Water Discharge 3-113
3.4.2.1 Data Sources 3-113
3.4.2.2 Quality and Quantity of Discharge 3-114
3.4.3 Hydraulic and Water Quality Effects of Underground Mine Discharge 3-120
3.4.3.1 Runoff and Flooding in the Model Underground Mine Area 3-120
3.4.3.1.1 Study Approach 3-120
3.4.3.1.2 Description of Area 3-122
3.4.3.1.3 Estimate of Sub-basin Flood Flow . . 3-124
3.4.3.1.4 Prediction of Sub-basin Water Quality . 3-132
3.4.3.2 Impacts of Seepage on Groundwater . . . 3-149
3.4.4 Gases and Dusts from Mining Activities . . . 3-155
3.4.4.1 Radon-222 in Mine Exhaust Air 3-155
3.4.4.2 Aboveground Radon-222 Sources 3-157
3.4.4.3 Dusts and Fumes 3-159
3.5 In Situ Leach Mining 3-168
3.5.1 Solid Wastes 3-169
3.5.2 Associated Wastewater 3-172
3.5.3 Airborne Emissions 3-174
3.5.4 Excursion of Lixiviant 3-178
3.5.5 Restoration and Reclamation 3-179
3.6 Other Sources 3-185
3.6.1 Mineral Exploration 3-185
3.6.1.1 Environmental Considerations 3-187
3.6.1.2 Radon Losses from Drill Holes 3-192
3.6.1.3 Groundwater 3-193
3.6.1.4 Fumes 3-193
3.6.1.5 Model Drilling 3-194
3.6.2 Precipitation Runoff from Uranium Mines . . . 3-194
3.7 Inactive Mines 3-204
3.7.1 Inactive Surface Mines 3-204
3.7.1.1 Waste Rock Piles 3-213

Page

3.7.1.2 Radon-222 from the Mine Area 3-216
3.7.1.3 Land Surface Gamma Radiation 3-222
3.7.2 Inactive Underground Mines 3-227
3.7.2.1 Waste Rock Piles 3-229
3.7.2.2 Radon-222 from the Mine Area 3-232
3.7.2.3 Land Surface Gamma Radiation 3-237
3.8 References 3-242
4.0 Description of Model Mines 4-1
4.1 Surface Mine 4-1
4.2 Underground Mine 4-4
4.3 In Situ Leach Mine 4-7
4.4 Inactive Surface Mine 4-8
4.5 Inactive Underground Mine 4-9
5.0 Potential Pathways 5-1
5.1 General 5-1
5.1.1 Vegetation 5-1
5.1.2 Wildlife 5-1
5.1.3 Land Use 5-2
5.1.4 Population Near Mining Areas 5-2
5.1.5 Population Statistics of Humans and Beef Cattle . 5-12
5.2 Prominent Environmental Pathways and Parameters for Aqueous Releases 5-12
5.2.1 Individual Committed Dose Equivalent Assessment . 5-13
5.2.2 Collective (Population) Dose Equivalent Assessment . 5-15
5.3 Prominent Environmental Pathways and Parameters for Atmospheric Releases 5-16
5.3.1 Individual Committed Dose Equivalent Assessment . 5-16
5.3.2 Collective (Population) Dose Equivalent Assessment . 5-18
5.4 Mine Wastes Used in the Construction of Habitable Structures 5-19
5.5 References 5-20
6.0 Health and Environmental Effects 6-1
6.1 Health Effects and Radiation Dosimetry 6-1

Page

6.1.1 Radioactive Airborne Emissions 6-1
6.1.2 Nonradioactive Airborne Emissions 6-26
6.1.2.1 Combustion Products 6-26
6.1.2.2 Nonradioactive Gases 6-28
6.1.2.3 Trace Metals and Particulates in the Form of Dust 6-28
6.1.3 Radioactive Aquatic Emissions 6-35
6.1.4 Nonradioactive Aquatic Emissions 6-42
6.1.5 Solid Wastes 6-44
6.1.5.1 Radium-226 Content 6-44
6.1.5.2 Estimates of Potential Risk 6-45
6.1.5.3 Using Radium Bearing Wastes in the Construction of Habitable Structures 6-46
6.1.5.3.1 Use of Uranium Mine Wastes . . . 6-48
6.2 Environmental Effects 6-48
6.2.1 General Considerations 6-48
6.2.2 Effects of Mine Dewatering 6-52
6.2.3 Erosion of Mined Lands and Associated Wastes . . 6-54
6.2.4 Land Disturbance from Exploratory and Development Drilling 6-56
6.2.5 Land Disturbance from Mining 6-59
6.2.5.1 Underground Mines 6-59
6.2.5.2 Surface Mines 6-59
6.2.6 Retirement Phase 6-59
6.3 References 6-73
7.0 Summary and Recommendations 7-1
7.1 Overview 7-1
7.2 Sources and Concentrations of Contaminants 7-1
7.2.1 Surface and Underground Mines 7-1
7.2.2 In Situ Leach Mines 7-7
7.2.3 Uranium Exploration 7-8
7.3 Exposure Pathways 7-8
7.4 Potential Health Effects 7-9

Page

7.4.1 Radioactive Airborne Emissions 7-9
7.4.2 Nonradioactive Airborne Emissions 7-13
7.4.3 Radioactive Aqueous Emissions 7-13
7.4.4 Nonradioactive Aqueous Emissions 7-15
7.4.5 Solid Wastes 7-16
7.5 Environmental Impacts 7-16
7.5.1 Land and Water Contamination 7-17
7.5.2 Effects of Mine Dewatering 7-22
7.5.3 Erosion of Mined Lands and Associated Wastes . . 7-22
7.5.4 Exploratory and Development Drilling . . . 7-23
7.5.5 Underground Mining 7-23
7.5.6 Surface Mining 7-24
7.6 Regulatory Perspective 7-25
7.7 Conclusions and Recommendations 7-25
7.7.1 Conclusions 7-26
7.7.1.1 Solid Wastes 7-26
7.7.1.2 Airborne Effluents 7-26
7.7.1.3 Waterborne Effluents 7-27
7.7.1.4 Exploratory and Development Drilling . . . 7-27
7.7.2 Recommendations to Congress 7-28
7.8 Other Findings 7-28
7.9 References 7-29

Appendixes (See Volume 3)

A. Summary of Federal Laws Potentially Affecting Uranium Mining
B. Federal Water Programs and Rights Activities
C. Congressionally Approved Compacts that Apportion Water
D. State Laws, Regulations, and Guides for Uranium Mining
E. Active Uranium Mines in the United States
F. Inactive Uranium Mines in the United States
G. General Observations of Uranium Mine Sites in Colorado, New Mexico, Texas, and Wyoming.

H. Influence of Mine Drainage on Seepage to Groundwater and Surface Water Outflow
I. Computation of Mass Emission Factors for Wind Erosion
J. Aquatic Dosimetry and Health Effects Models and Parameter Values
K. Airborne Pathway Modeling
L. Health Risk Assessment Methodology

FIGURES

Page

1.1 Uranium mining regions in the western United States . . 1-4

1.2 The percent of $50 U_3O_8 reserves located in the principal mining states 1-8

1.3 The percent of $50 U_3O_8 reserves located in the various mining regions 1-9

1.4 Artist's conception of open pit mining operation and support facilities 1-13

1.5 Generalized underground mine showing modified room and pillar method of mining 1-17

1.6 Diagrams of some common injection-recovery well patterns used in uranium in situ leach mining 1-26

2.1 Location of active and inactive uranium mines and principal uranium mining districts in Colorado 2-6

2.2 Location of active and inactive uranium mines and principal uranium mining districts in the Uravan Mineral Belt of western Colorado 2-7

2.3 Location of active and inactive uranium mines in the Grants Mineral Belt and other areas of New Mexico 2-8

2.4 Location of active, inactive, and proposed surface and in situ uranium mines in Texas 2-9

2.5 Location of uranium mines and mining districts in Utah . . 2-10

2.6 Location of uranium mines and principal uranium mining districts in southeastern Utah 2-11

2.7 Location of active and inactive uranium mines and principal uranium mining areas in Wyoming 2-12

2.8 Location of active and inactive uranium mines in the Gas Hills and Crooks Gap-Green Mountain areas of central Wyoming . . 2-13

2.9 Location of active and inactive uranium mines in the Shirley Basin, South Powder River Basin, and Pumpkin Buttes areas of Wyoming 2-14

Page

3.1 The uranium decay series showing the half lives and mode of decay 3-2
3.2 The thorium decay series showing the half lives and mode of decay 3-3
3.3 Disposition of drainage water from active surface and underground uranium mines 3-14
3.4 Location of mines, ore and waste storage areas and monitoring stations at the Morton Ranch mine, South Powder River Basin, Wyoming 3-37
3.5 Location of study areas, sampling stations and uranium mines, Poison Canyon area, McKinley County, New Mexico . . . 3-41
3.6 Sample locations for radionuclides and select trace metals in sediments, San Mateo mine, New Mexico 3-42
3.7 Potential sources of environmental contamination from active open pit uranium mines 3-46
3.8 Storage pile configurations assumed at surface and underground mines 3-48
3.9 Sketch of sub-basin, basin, and regional basin showing orientation of principal drainage courses, areas of drainage, and location of mines 3-69
3.10 Average monthly flows for the Cheyenne River and Lance Creek near Spencer, Wyoming, for the period 1948-1970 . . . 3-71
3.11 Suspended sediment concentration to discharge, Salt Wells Creek and Tributaries, Wyoming 3-75
3.12 Relation of discharge and specific conductance to time at Salt Wells Creek, Green River Basin, Wyoming 3-76
3.13 Periods of no flow in Lance Creek and the Cheyenne River near Riverton, Wyoming, for the period 1948-1978. 3-82
3.14 Configuration of open pit model mines 3-105
3.15 Potential sources of environmental contamination from active underground uranium mines 3-108

Page

3.16 Sketch of sub-basin, basin, and regional basin showing orientation of principal drainage courses, areas of drainage, and location of mines in the New Mexico model area . 3-121
3.17 Average monthly flows for the period of record for the Rio San Jose and the Rio Puerco in New Mexico 3-129
3.18 Periods of no flow in the Rio San Jose and Rio Puerco . . 3-131
3.19 Total flow volumes in one-day periods for floods of various recurrence intervals in the sub-basin and basins in New Mexico 3-133
3.20 Total flow volumes in seven-day periods for floods of various recurrence intervals in the sub-basin and basins in New Mexico 3-134
3.21 Principal streams and surface water sampling stations in the Churchrock and Gallup areas 3-145
3.22 Average depth of exploratory drilling in the U.S. uranium industry from 1948 to present 3-188
3.23 Annual waste to ore ratios for surface mining of uranium (1948 to 1979) 3-209
3.24 Cross section of model inactive surface mine . . . 3-214
3.25 Results of gamma exposure rate survey at the 1601 pit and environs, Morton Ranch uranium mine, Converse County, Wyoming 3-226
3.26 Waste to ore ratios for inactive underground uranium mines from 1932 to 1977 3-228
3.27 Radon-222 concentrations in mine air discharged by natural ventilation 3-236
3.28 Gamma radiation survey around an inactive underground uranium mine in New Mexico 3-240
5.1 Potential airborne pathways in the vicinity of uranium mines 5-17

Page

6.1 Average indoor radon-222 decay product measurements (in working levels) as a function of average radium-226 concentration in soil 6-47
6.2 Example of natural reclamation of drill sites . . . 6-57
6.3 Inactive underground mine site 6-60
6.4 Example of active and inactive surface mining activities . 6-63
6.5 Mine wastes eroded by ephemeral streams in the Mesa Montanosa area, New Mexico 6-64
6.6 Basal erosion of a uranium mine waste pile by an ephemeral stream in the Mesa Montanosa area, New Mexico . . . 6-65
6.7 Scattered piles of mine waste at the Mesa Top mine, Mesa Montanosa, New Mexico 6-66
6.8 Close up view of easily eroded sandy and silty mine waste from the Mesa Top mine, Mesa Montanosa, New Mexico . . 6-66
6.9 Gullying and sheet erosion of piled and spread mine wastes at the Dog Incline uranium mine, Mesa Montanosa, New Mexico . 6-67
6.10 Recent erosion of unstabilized overburden piles at the inactive Galen mine, Karnes County, Texas 6-68
6.11 Unstabilizied overburden piles and surface water erosion at the Galen mine, Karnes County, Texas 6-68
6.12 Aerial view of the Manka mine, Karnes County, Texas . . 6-70
6.13 Overburden pile showing the weak vegetative cover and gullying associated with improper stabilization at the Manka mine, Karnes County, Texas, ... 6-70
6.14 Inactive Hackney mine, Karnes County, Texas 6-72

Appendix G

G.1 Plan view of inactive underground uranium mine No. 1, related waste rock piles, and surface gamma exposure rates, Uravan Mineral Belt, Colorado G-3

Page

G.2 Sectional view of inactive underground uranium mine No. 2, related waste rock piles, and surface gamma exposure rates, Uravan Mineral Belt, Colorado G-4

G.3 Plan view of inactive underground uranium mine No. 3, related waste rock piles, and surface gamma exposure rates, Uravan Mineral Belt, Colorado G-6

G.4 Sectional view of inactive underground uranium mine No. 4, related waste rock piles, and surface gamma exposure rates, Uravan Mineral Belt, Colorado G-7

G.5 Plan view of inactive underground uranium mine No. 5, related waste rock piles, and surface gamma exposure rates, Uravan Mineral Belt, Colorado G-8

G.6 Plan view of inactive underground uranium mine No. 6, related waste rock piles, and surface gamma exposure rates, Uravan Mineral Belt, Colorado G-9

G.7 Plan view of inactive underground uranium mine No. 7, related waste rock piles, and surface gamma exposure rates, Central City District, Colorado G-11

G.8 Plan view of inactive underground uranium mine No. 8, related waste rock piles, and surface gamma exposure rates, Central City District, Colorado G-12

G.9 Plan view of inactive underground fluorspar uranium mine No. 9, related waste rock piles, and surface gamma exposure rates, near Jamestown, Colorado G-13

G.10 Plan view of inactive underground uranium mine No. 10, related waste rock piles, and surface gamma exposure rates, Central City District, Colorado G-15

G.11 Typical mine waste pile associated with a small- to medium-sized inactive underground uranium mine, Uravan Mineral Belt, Colorado G-16

G.12 Side view of a typical underground uranium mine located on the rim of a sandstone mesa, Uravan Mineral Belt, Colorado . G-16

Page

G.13 Mine waste accumulations near the portal of a typical underground rim-type uranium mine, western Colorado . . . G-17

G.14 Mine waste dump associated with a typical rim-type underground uranium mine, western Colorado G-17

G.15 Movement of fluorspar-uranium mine wastes from a tailings pile into a stream, Jamestown area, Colorado G-19

G.16 1972 aerial photograph of the Galen and Pawelek open pit mines, Karnes County, Texas G-27

G.17 1978 aerial photograph of the Galen and Pawelek open pit mines, Karnes County, Texas G-27

G.18 Results of gamma exposure rate survey at the 1601 pit and environs, Morton Ranch uranium mine, Converse County, Wyoming G-31

G.19 Location of sampling stations at the Morton Ranch mine, South Powder River Basin, Wyoming G-33

G.20 Sample locations for radionuclides and select trace metals in sediments, San Mateo mine, New Mexico G-34

Appendix H

H.1 Wyoming model area sub-basin drainage system H-3

H.2 Model area stream cross section H-3

H.3 New Mexico model area sub-basin drainage system . . . H-9

Appendix J

J.1 Surface stream flow pattern within drainage area . . . J-3

J.2 Conservation of mass relationship for resuspension model . J-13

TABLES

Page

1.1 Domestic uranium production by state from 1948 to January 1, 1979 1-3
1.2 Projected annual nuclear capacity (GWe) in the U.S. . 1-6
1.3 Domestic uranium reserves by state as of January 1, 1979 1-7
1.4 The quantities of U_3O_8 produced in 1978 by the various mining methods 1-10
1.5 The predicted methods of mining ore reserves . . . 1-10
1.6 Summary of current in situ leaching operations as of January 1, 1978 1-21
1.7 Trace metal concentrations of recirculated acid and alkaline lixiviants 1-25
1.8 Federal laws, regulations, and guides for uranium mining 1-32
1.9 Requirements to obtain rights to prospect or explore by federal, state and private lands 1-37
1.10 Requirements to obtain rights to mine ore by federal, state and private lands 1-39
1.11 Requirements for mining and environmental plans by federal, state and private lands 1-41
1.12 State laws, regulations, and guides for uranium mining . 1-43
2.1 Type of U.S. uranium properties 2-3
2.2 The location and type of active uranium properties . 2-4
2.3 The location and type of inactive uranium properties . 2-5
2.4 Cumulative ore production through January 1, 1979 . . 2-15
3.1 Gamma-ray energy released by one gram of rock . . 3-4
3.2 Radionuclide content and dose equivalent rates from common rocks and soil 3-4
3.3 Average dose equivalent rates due to terrestrial radiation in western mining states 3-5
3.4 Radionuclide concentrations in surface and groundwater in the vicinity of a proposed uranium project . . 3-7
3.5 Concentrations of selected elements in igneous and sedimentary rocks 3-8

Page

3.6 Concentrations of selected elements in surface water at five locations in the vicinity of a proposed uranium project 3-9
3.7 Concentrations of selected elements in groundwater at six locations in the vicinity of a proposed uranium project. 3-10
3.8 Estimated average concentrations of three metals in U.S. streams 3-6
3.9 Summary of feed water sources for active U.S. uranium mills 3-17
3.10 Current and projected uranium mine discharges in the Grants Mineral Belt, New Mexico 3-21
3.11 Estimated surface areas associated with overburden piles 3-50
3.12 Particle size distributions of mill tailings and mine overburden 3-52
3.13 Natural radionuclide concentrations in various common rock types 3-52
3.14 Annual average airborne radionuclide concentrations in the vicinity of an open pit uranium mine . . . 3-53
3.15 Uranium and stable element concentrations measured in rock and soil samples from two uranium mines 3-55
3.16 Concentration of radionuclides and stable elements in overburden rock from the model surface mines . . . 3-56
3.17 Estimated average areas of ore pile surface and pad . 3-57
3.18 Distribution of ore reserves by the type of host . . 3-59
3.19 Average stable element concentrations in sandstone ores of New Mexico 3-59
3.20 Estimated average surface areas of sub-ore piles during the 17-year active mining period 3-61
3.21 Summary of average discharge and water quality data for uranium mines in Wyoming and a comparison with NPDES limits 3-65
3.22 Water quality associated with surface and underground mines in various stages of construction and operation . 3-67

Page

3.23 Peak discharge and total volume for floods of 2, 5, 10, 25, 50 and 100 year recurrence intervals 3-78
3.24 Summary of calculated total flow in the Wyoming model area sub-basin using the USGS and SCS methods . . 3-79
3.25 Annual contaminant loading from one uranium mine and resulting concentrations in floods within the sub-basin for return periods of 2 to 100 years 3-84
3.26 Concentrations in basin and regional basin streams as a result of surface mine discharge 3-86
3.27 Comparison of potable and irrigation water standards and surface water quality affected by surface mine drainage 3-87
3.28 Northeastern Wyoming groundwater sources . . . 3-91
3.29 Groundwater quality of wells sampled by the three major uranium producers in the South Powder River Basin, Wyoming 3-92
3.30 Estimated air pollutant emissions from heavy-duty equipment at surface mines 3-94
3.31 Average annual dust emissions from mining activities . 3-97
3.32 Average annual emissions of radionuclides (μCi) and stable elements (kg) from vehicular dust at the model surface mines 3-100
3.33 Average annual emissions of radionuclides (μCi) and stable elements (kg) from mining activities at the model surface mines 3-101
3.34 Average annual emissions of radionuclides (μCi) and stable elements (kg) in wind suspended dust at the model surface mines 3-102
3.35 Radon-222 releases during surface mining . . . 3-107
3.36 Estimated average surface areas of waste rock piles at underground mines 3-111
3.37 Estimated surface areas of ore stockpiles at underground mines 3-111
3.38 Estimated average surface areas of sub-ore piles at underground mines 3-113

Page

3.39 Summary of average discharge and water quality data for underground uranium mines in the Colorado Plateau Region (Colorado, New Mexico, Utah) and a comparison with NPDES limits 3-116

3.40 Water quality associated with underground mines in various stages of construction and operation . . . 3-119

3.41 Total flow volume for sub-basin floods of 1- and 7-day durations and return periods of 2, 5, 10, and 25 years . 3-126

3.42 Summary of area, discharge, and irrigated acreage for the sub-basin, basin, and regional basin hydrographic units in New Mexico 3-127

3.43 Dilution factors for the Rio San Jose, Rio Puerco, and Rio Grande for 1-day flood flows with a 2-year recurrence interval 3-135

3.44 Annual contaminant loading from 14 uranium mines and resulting concentrations in sub-basin floods and in the average annual flow of the Rio San Jose, Rio Puerco, and Rio Grande 3-137

3.45 Comparison of potable and irrigation water standards and surface water quality affected by underground mine drainage. 3-140

3.46 Radiochemical and stable element/compound water quality for selected acid and alkaline leach uranium mill tailings ponds in the United States 3-142

3.47 Summary of flood runoff water quality and uranium millpond quality 3-143

3.48 Flow and water quality in the Puerco River near Churchrock and Gallup, New Mexico 3-146

3.49 Groundwater quality in principal aquifers in the Grants Mineral Belt, New Mexico 3-151

3.50 Groundwater quality associated with the San Mateo Creek and Rio Puerco (west) drainages in the Grants Mineral Belt, New Mexico 3-154

Page

3.51 Estimated annual radon-222 emissions from underground uranium mining sources 3-158

3.52 Estimated air pollutant emissions from heavy-duty equipment at underground uranium mines 3-161

3.53 Estimated average annual dust emissions from underground mining activities 3-163

3.54 Average annual emissions of radionuclides (μ Ci) and stable elements (kg) from mining activities at the model underground mines. 3-165

3.55 Average annual emissions of radionuclides (μ Ci) and stable elements (kg) in wind suspended dust at the model underground mines 3-166

3.56 Average annual emissions of radionuclides (μ Ci) and stable elements (kg) from vehicular dust at the model underground mines 3-167

3.57 Estimated quantities of wastewater produced by an in situ leaching operation 3-173

3.58 Estimated average concentrations and annual accumulation of some contaminants in waste water 3-175

3.59 Estimated average annual airborne emissions from the hypothetical in situ leaching facility 3-176

3.60 Estimated average concentrations and annual and total accumulations of some contaminants in restoration wastewater 3-181

3.61 A comparison of contaminant concentrations in pre-mining groundwater and pre-restoration mine water 3-182

3.62 Estimates of exploratory and development drill holes (1948-1979) 3-189

3.63 Estimated source terms per bore hole for contemporary surface drilling for uranium 3-195

3.64 Airborne dusts produced at an average mine site from exploratory and development drilling 3-196

Page

3.65 Estimates of emissions from drill rig diesel power source 3-196
3.66 Sediment yields in overland flow from uranium mining areas 3-202
3.67 Consolidated list of inactive uranium producers by State and type of mining 3-206
3.68 Uranium mine waste and ore production 3-207
3.69 Cumulative uranium mine waste and ore production . . 3-211
3.70 Average annual emissions of radionuclides (μCi) and stable elements (kg) in wind suspended dust at the model inactive surface mine 3-217
3.71 Average radon flux of inactive uranium mill tailings piles 3-218
3.72 Average radon flux measured at inactive uranium mine sites 3-220
3.73 Background radon flux estimates 3-221
3.74 Summary of estimated radon-222 releases from inactive surface mines 3-223
3.75 Summary of land surface gamma radiation surveys in New Mexico, Texas, and Wyoming 3-225
3.76 Average annual emissions of radionuclides (μCi) and stable elements (kg) in wind suspended dust at the model inactive underground mine 3-231
3.77 Summary of radon-222 releases from inactive underground mines 3-238
3.78 Summary of land surface gamma radiation surveys in Colorado and New Mexico 3-239
5.1 Number of uranium mines and population statistics for counties containing uranium mines. 5-3
5.2 Population statistics for humans and beef cattle . . 5-12
5.3 Aquatic environmental transport pathways initially considered 5-14

Page

6.1 Annual release rates (Ci) used in the dose equivalent and health effects computations for active uranium mines 6-2
6.2 Annual release rates (Ci) used in the dose equivalent and health effects computations for inactive uranium mines 6-4
6.3 Annual working level exposure from radon-222 emissions from model uranium mines 6-6
6.4 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model average surface uranium mine 6-7
6.5 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model average large surface uranium mine 6-8
6.6 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model average underground uranium mine 6-9
6.7 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model average large underground uranium mine 6-10
6.8 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model inactive surface uranium mine 6-11
6.9 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model inactive underground uranium mine 6-12
6.10 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a hypothetical in situ uranium solution mine 6-13
6.11 Individual lifetime fatal cancer risk for one year of exposure and estimated additional fatal cancers to the regional population due to annual radioactive airborne emissions from model uranium mines 6-15

Page

6.12 Individual lifetime fatal cancer risk due to lifetime exposure to radioactive airborne emissions from model uranium mines 6-16

6.13 Genetic effect risk to descendants for one year of parental exposure to atmospheric radioactive airborne emissions from model uranium mines 6-17

6.14 Genetic effect risk to descendants for a 30-year parental exposure to atmospheric radioactive airborne emissions from model uranium mines 6-18

6.15 Percent of the fatal cancer risk for the maximum individual due to the sources of radioactive emissions at model uranium mines 6-20

6.16 Percent of the fatal cancer risk for the average individual in the regional population due to the sources of radioactive emissions at model uranium mines 6-21

6.17 Percent of fatal cancer risks due to radon-222 daughter concentrations at model uranium mine sites 6-22

6.18 Percent of the fatal cancer risk for principal nuclides and pathways due to radioactive particulate and Rn-222 emissions at model uranium mines 6-23

6.19 Natural background concentrations and average urban concentrations of selected airborne pollutants in the United States 6-27

6.20 Combustion product concentrations at the site of the maximum individual with comparisons 6-29

6.21 A comparison of the airborne concentrations of nonradioactive gases at the hypothetical in situ leach site with threshold limit values 6-30

6.22 Stable trace metal airborne concentrations at the site of the maximum individual 6-32

6.23 Comparison of stable trace metal airborne concentrations at the location of the maximum individual with natural background concentrations and average urban concentrations of these airborne pollutants 6-33

Page

6.24 Comparison of trace metal airborne concentrations at the site of the maximum individual with threshold limit values (TLV's) in the workroom environment adjusted for continuous exposure to the general public 6-34

6.25 Annual radiation dose equivalent rates due to aquatic releases from the New Mexico model underground mine . 6-36

6.26 Annual radiation dose equivalent rates due to aquatic releases from the Wyoming model surface mine 6-37

6.27 Individual lifetime fatal cancer risk and committed fatal cancers to the population residing within the assessment areas 6-38

6.28 Genetic risks to succeeding generations of an individual and committed genetic effects to descendants of the present population residing within the assessment area . . 6-41

6.29 Comparison of nonradiological waterborne emissions from uranium mines with recommended agricultural water quality limits 6-43

6.30 Estimated lifetime risk of fatal lung cancer to individuals living in homes built on land contaminated by uranium mine wastes 6-45

6.31 Gamma radiation anomalies and causes 6-49

7.1 Distribution of United States uranium mines by type of mine and state 7-2

7.2 Sources of contaminants at uranium mines . . . 7-3

7.3 Concentration of contaminants in waste rock (overburden), ore, and sub-ore 7-6

7.4 Summary of harm from radioactive airborne emissions of model uranium mines 7-11

7.5 Percent additional lifetime fatal cancer risk for a lifetime exposure to the individual and the percent additional cancer deaths in the regional population per year of exposure estimated to occur as a result of uranium mining 7-12

Page

7.6 Summary of the fatal cancer risks caused by radioactive aqueous emissions from model uranium mines . . . 7-14
7.7 Estimated lifetime risk of fatal lung cancer to the average person living in a home built on land contaminated by uranium mine wastes 7-16
7.8 Summary of contaminant loading and stream water quality from a model surface uranium mine 7-20
7.9 Summary of contaminant loading and stream water quality from a model underground uranium mine 7-21

Appendix A

A.1 Federal laws, regulations, and guides for uranium mining A-1

Appendix D

D.1 State laws, regulations, and guides for uranium mining . D-1

Appendix E

E.1 Active uranium mines in the United States . . . E-1

Appendix F

F.1 Inactive uranium mines in the United States . . . F-1

Appendix G

G.1 Uravan and Jamestown areas G-14
G.2 Inactive uranium mine sites surveyed in New Mexico . G-21
G.3 Status and location of uranium mines in Texas. . . . G-25
G.4 Trace elements and radionuclides in water in the south fork of Box Creek drainage at UNC Morton Ranch lease . G-35
G.5 Radionuclides and trace metals in sediments in the south fork of Box Creek at UNC Morton Ranch lease . . . G-36
G.6 Radionuclides and trace metals in soils near the 1601 open pit mine, UNC Morton Ranch lease, Wyoming . . G-37

Page

G.7 Radionuclides and trace metals in soil profiles at the open pit mines, UNC Morton Ranch lease, Wyoming . . G-38
G.8 Radionuclides and trace metals in sediments from the drainage of the San Mateo mine and from San Mateo Creek, New Mexico G-40
G.9 Radium-226 and trace elements in water from San Mateo Creek near San Mateo mine discharge point . . . G-40

Appendix H

H.1 Characteristics of the sub-basin containing the model mines H-2
H.2 Seepage and outflow calculations for the Wyoming model mine drainage system H-6
H.3 Characteristics of the sub-basin hydrographic unit in the model underground uranium mine area H-8
H.4 Seepage and outflow calculations for the New Mexico model mine area drainage system H-11

Appendix J

J.1 Aquatic environmental transport pathways examined . . J-6
J.2 Characteristics of the generic sites J-20
J.3 Stream data for Valencia County J-23
J.4 Estimation of meat production in Valencia County for 1977 . . . . J-25
J.5 Estimates of meat production in Converse County, Wyoming for 1976 . . . . J-26
J.6 Annual radionuclide release rates to streams for active uranium mines J-29
J.7 Freshwater fish concentration factors J-29
J.8 Normalized human intake rate factors for radionuclide uptake via plant root systems J-31
J.9 Irrigated land usage J-31

Page

J.10 Soil removal rate constants and radioactive decay constants J-33
J.11 Milk and beef concentration factors J-33
J.12 Dose equivalent conversion factors J-36
J.13 Health effects conversion factors for internal pathways. J-37
J.14 Health effects conversion factors for external pathways. J-37

Appendix K

K.1 Characteristics of the generic sites K-1
K.2 Animal and vegetable crop distribution for use with AIRDOS-EPA K-3
K.3 Sources of food for the maximum individual (percent) . K-4
K.4 Selected input parameters to AIRDOS-EPA K-5
K.5 Selected terrestrial pathway parameters by radionuclide. K-7
K.6 Effective radioactive decay constants K-8

Appendix L

L.1 Radionuclide dose rate and health effect risk conversion factors used in uranium mine assessments L-4
L.2 Additional input data used by DARTAB in the health impact assessment of airborne emissions L-21
L.3 Example input data file for DARTAB L-22
L.4 Maximum individual fatal cancer risk for one year of exposure to atmospheric radioactive emissions from model uranium mines L-23
L.5 Fatal cancer risk to an average individual in the regional population for one year of exposure to atmospheric radioactive emissions from model uranium mines L-24
L.6 Fatal cancer risk to the population for one year of exposure to atmospheric radioactive emissions from model uranium mines L-25

Page

L.7 Genetic effect risk to descendants of maximum exposed individual for one year of parental exposure to atmospheric radioactive particulate and Rn-222 emissions from model uranium mines L-26
L.8 Genetic effect risk to descendants of average individual of the population for one year of parental exposure to atmospheric radioactive particulate and Rn-222 emissions from model uranium mines L-27
L.9 Genetic effect risk to descendants of the regional population for one year of parental exposure to atmospheric radioactive particulates and Rn-222 emissions from model uranium mines L-28

FOREWORD

The Uranium Mill Tailings Radiation Control Act of 1978 stipulates that "...the Administrator, in consultation with the Commission, shall provide to the Congress a report which identifies the location and potential health, safety, and environmental hazards of uranium mine wastes together with recommendations, if any, for a program to eliminate these hazards." It is our understanding that the intent of Congress was to determine if remedial actions similar to those for uranium mill tailings are required for mine wastes.

The report was prepared by the Office of Radiation Programs and addresses potential health effects caused by air emissions, water effluents, and solid wastes at active and inactive uranium mines. It is probably the single most comprehensive report on the subject. The effects from other mining activities such as exploration, site preparation, and in situ leaching were evaluated in proportion to their potential significance and the amount of available information about them. Comments on this report from the uranium mining industry, States, and the Nuclear Regulatory Commission have also been considered.

The conclusions and recommendations are in the Executive Summary and Chapter 7 of the report. The principal findings of this report are as follows:

1. No problems were identified that require Congressional action.

2. Standards are probably needed to control human exposure from radioactive emissions from underground uranium mines. We have proposed a standard for underground uranium mines under the Clean Air Act program to develop radionuclide standards.

3. Regulations should be considered for maintaining the control of solid wastes at active uranium mines to prevent off-site use and to minimize the health risks from these materials. This is part of the overall agency consideration of mining wastes and is being carried out under the auspices of the Solid Waste Disposal Act.

4. The report also identifies additional studies that are needed to completely elucidate the potential for local adverse effects as a result of possible misuse of the mine waste materials in construction of buildings. Preliminary reports of field studies by EPA identifying possible sites at which mine wastes may have been utilized in construction or around buildings have been sent to the States for follow-up studies.

Glen L Sjoblom

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SECTION 1

INTRODUCTION

1.0 Introduction

1.10 Purpose

This report was prepared in response to Section 114(c) of Public Law 95-604 dated November 8, 1978 (USC78). This Section of the Law stipulates that, "Not later than January 1, 1980, the Administrator, in consultation with the Commission, shall provide to the Congress a report which identifies the location and potential health, safety, and environmental hazards of uranium mine wastes together with recommendations, if any, for a program to eliminate these hazards." The purpose of this report is to comply fully with this request, as accurately and completely as available information will permit.

1.1.1 Contents

This volume has seven major sections. The content of each section is described generally below:

Section 1: Brief reviews of predicted future uranium production requirements; descriptions of methods of extracting uranium from the earth; and presently enforced standards and regulations governing uranium mining.

Section 2: A description of the active and inactive uranium mine inventory with a discussion of its limitations. The actual mine listings are presented in Appendixes E and F.

Section 3: A comprehensive discussion of potential sources of radioactive and stable contaminants to the environment and man from uranium mining operations. Annual release rates of contaminants from the identified sources computed on a generic basis.

Section 4: A description of model underground, surface, and in situ leach mines with operational parameters and source terms. Both active and inactive model underground and surface mines are described.

Section 5: A brief and general discussion of the environment that exists about uranium mines, including vegetation, wildlife, domestic animals, and human populations. The potential atmospheric and aquatic pathways of contaminants from the mines to man are also defined.

Section 6: Computation of individual and population dose equivalents and potential health effects from mine wastes and effluents based on the source terms developed in Section 3, the model mines defined in Section 4, and the pathways described in Section 5. A qualitative description of the environmental effects based on site visits is also presented.

Section 7: A brief summary of the report followed by the conclusions and recommendations.

1.2 Uranium Ore Production and Future Uranium Needs

1.2.1 Past Production

Table 1.1 lists the quantities of ore mined and uranium (U_3O_8) produced in the various uranium mining states between 1948 and January 1, 1979. Two states, New Mexico and Wyoming, have been the source of about 64 percent of the uranium mined in the United States. The Colorado Plateau, which includes parts of New Mexico, Arizona, Colorado, and Utah (see Figure 1.1), has been the largest source area of mined uranium, accounting for about 70 percent of the U_3O_8 production through 1976 (ST78). During this same period, the Gas Hills and the Shirley and Powder River Basins of Wyoming produced about 22 percent of the total U_3O_8 (ST78).

To produce 302,370 MT of U_3O_8 required the mining of 145,811,000 MT of uranium ore during the 31-year period from 1948 to 1979(DOE79). The average grade of ore, reported as percent of U_3O_8, was 0.208 percent during this period.

1.2.2 Projected Needs for Uranium

The expected growth in the use of nuclear energy for the production of electric power in the United States during the remainder of this century will require an expansion of the uranium mining industry. However, the magnitude of this expansion is difficult to estimate, because the forecasts that predict the growth of the nuclear power industry differ considerably (AEC74, ERDA75, EPA76, NRC76, NUS76, Cu77, He77, NEP77, Ew78, Ni78, and St78). However, all forecasts predict a continued growth of the industry. Expansion of the uranium mining industry will be influenced also by decisions regarding fuel reprocessing and commercial utilization of breeder reactors.

Table 1.1 Domestic uranium production by state from 1948 to January 1, 1979 (DOE79)

State	Ore Production, MT	Contained U_3O_8, MT[a]
New Mexico	59,977,000	123,560
Wyoming	36,263,000	68,400
Others[b]	49,571,000	110,410
Total	145,811,000	302,370

(a) Includes minor, non-ore sources.

(b) Includes Alaska, Arizona, California, Colorado, Florida, Idaho, Montana, Nevada, New Jersey, North Dakota, Oklahoma, Oregon, South Dakota, Texas, Utah, and Washington.

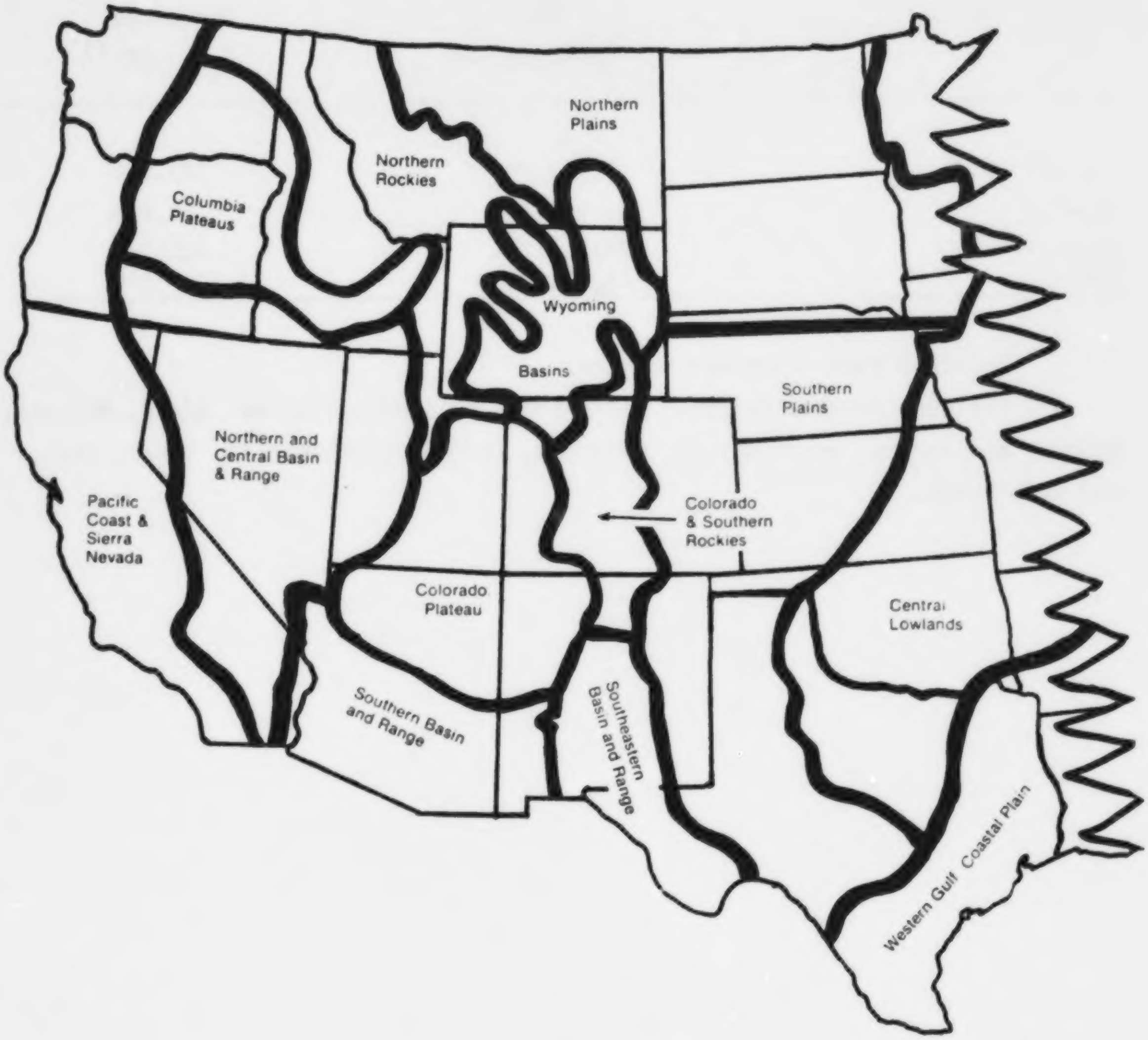

Northern Plains
Northern Rockies
Columbia Plateaus
Wyoming
Basins
Southern Plains
Northern and Central Basin & Range
Pacific Coast & Sierra Nevada
Colorado & Southern Rockies
Colorado Plateau
Central Lowlands
Southern Basin and Range
Southeastern Basin and Range
Western Gulf Coastal Plain

Figure 1.1 Uranium mining regions in the western United States

Table 1.2 gives examples of four typical forecasts. The Nuclear Regulatory Commission's (NRC) projected annual nuclear capacity is far below former predictions. This lower projection, which is in line with the administration's National Energy Plan (NEP77), is believed to be more realistic in view of recent drops in demand for electricity, labor problems, equipment delays, litigations initiated by environmental groups, the absence of a publicly accepted waste disposal program, and concern over nuclear proliferation. The Department of Energy predicts that 293,120 MT of U_3O_8 will be required to provide nuclear generating capacity through 1990 (DOE79). This assumes no uranium or plutonium recycling.

Table 1.3, which gives domestic uranium reserves by state, shows that the reserves are near the areas already mined. Figure 1.2 shows the distribution of \$50 ore reserves by state, and Fig. 1.3 shows reserves by resource region (see Fig. 1.1 for region locations). Future major mining activities probably will be in the same general areas that have already been mined. To obtain the 834,600 MT of \$50 U_3O_8 reserves will require mining about 1.14×10^9 MT of ore with an average grade of 0.073 percent U_3O_8.

1.3 Overview of Uranium Mining Operations

1.3.1 General

The two major uranium mining methods used in the United States are underground mining and surface (open pit) mining. These two methods accounted for more than 98 percent of the uranium mined in the United States in 1971 (AEC74). This has decreased only slightly to about 93 percent in 1978 (DOE79). However, various types of solution mining are currently being tested and probably will be employed commercially more frequently.

Table 1.4 shows the current production capacities of U_3O_8 for the various mining methods. Although underground mines are far more numerous than surface mines, production by the two methods is nearly equal. This is because surface mines have a much larger capacity. During 1978, 305 underground mines accounted for about 46 percent (8,350 MT) of the U_3O_8 production while 63 surface mines produced about 47 percent (8,710 MT) of the U_3O_8. In situ leaching, heap leaching, mine water extraction, and other alternative methods accounted for the remaining 7 percent (1,270 MT).

Table 1.2 Projected annual nuclear capacity (GWe) in the U.S.

Source	1980	1985	1990	1995	2000
ERDA (ERDA75)	71-92	160-245	285-470	445-790	625-1250
USEPA (EPA76)	80	188	350	578	820
Electric World (EW78)	92	160	194	237	---
NRC (NRC79)(a)	61	127	195	280	380

(a) Schedule assumed for this document.

Note.--The actual nuclear capacity realized in 1977 was 49 GWe.

Table 1.3 Domestic uranium reserves by state
as of January 1, 1979

State	Ore, MT	Ore Grade, % U_3O_8	U_3O_8, MT	% Total U_3O_8
New Mexico	482,200,000	0.09	434,000	52
Wyoming	431,300,000	0.06	258,800	31
Texas	83,400,000	0.05	41,700	5
Arizona, Colorado, & Utah	107,300,000	0.07	75,100	9
Others[a]	35,700,000	0.07	25,000	3
Total	1,139,900,000	0.073[b]	834,600	100

[a] Includes Alaska, California, Idaho, Montana, Nevada, North Dakota, Oregon, South Dakota and Washington.

[b] Weighted average.

Note.--The uranium reserves in this table include ore from which U_3O_8 can be obtained at a forward cost of $50 per pound or less. Costs do not include profits or cost of money.

Source: DOE79.

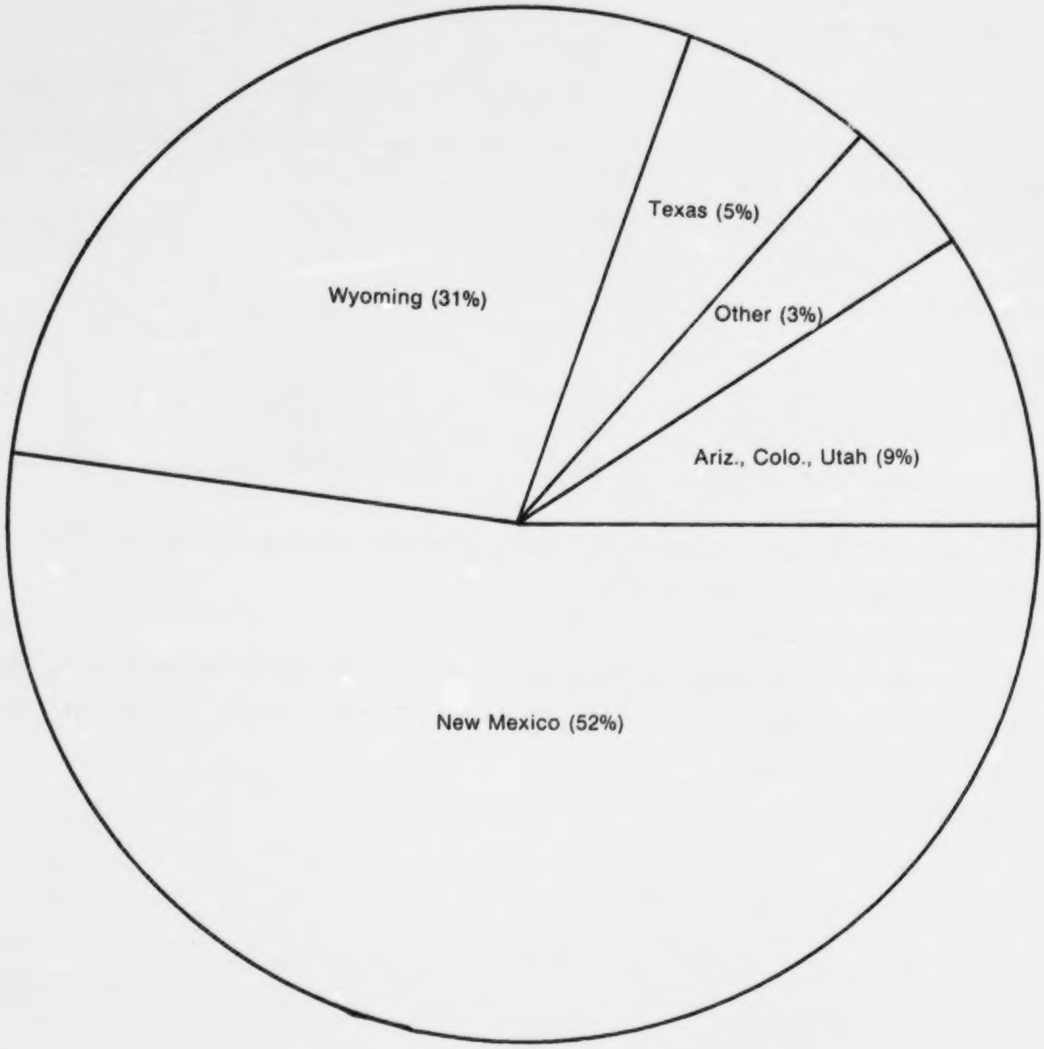


Figure 1.2 The percent of \$50 U_3O_8 reserves located in the principal mining states (DOE 79)

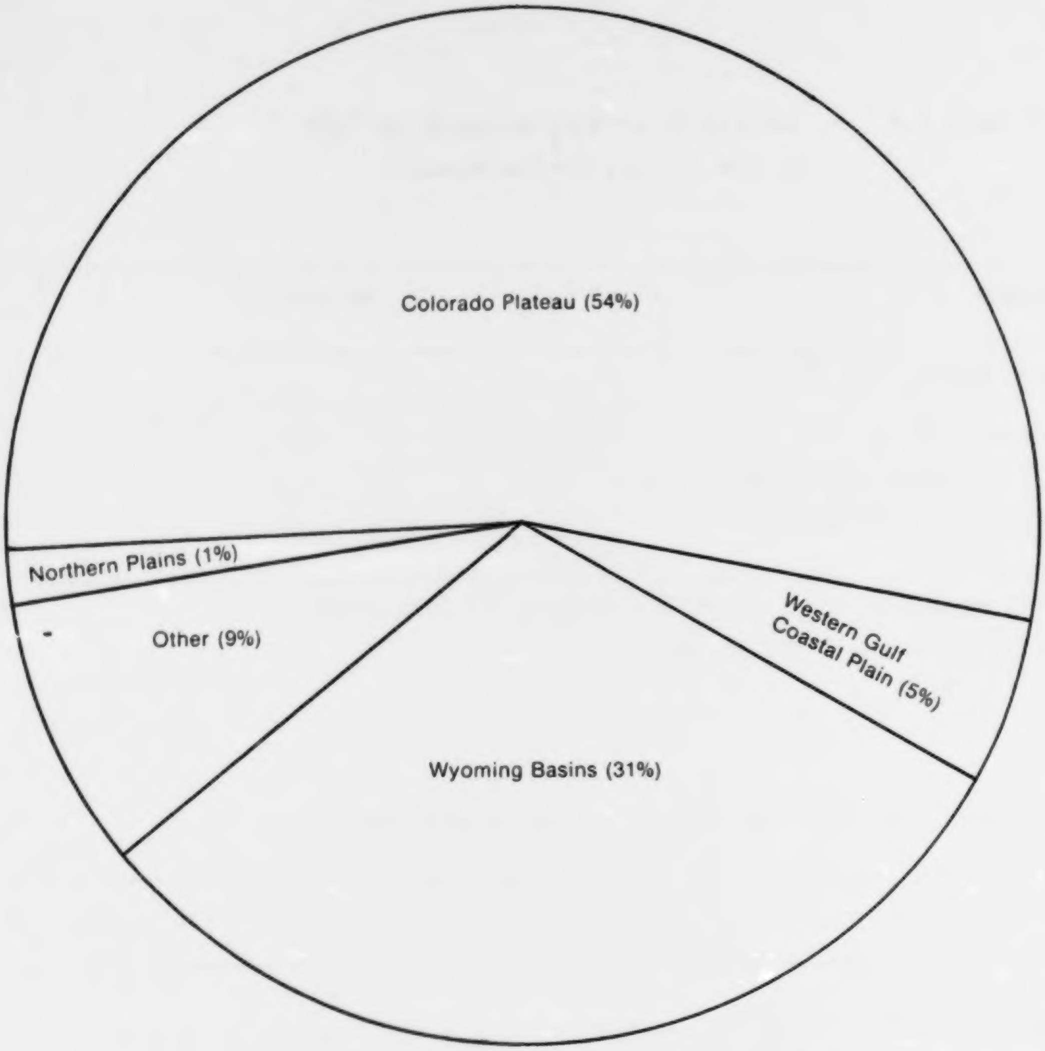


Figure 1.3 The percent of \$50 U_3O_8 reserves located in the various mining regions (DOE 79)

Table 1.4 Quantities of U_3O_8 produced in 1978 by the various mining methods

Mining Method	MT of U_3O_8	Percent[b]
Underground Mines (305)[a]	8350	46
Surface Mines (63)	8710	47
Other (23): In situ leaching, heap leaching, & mine water	1270	7
Total	18,330	100

[a] The number of mines or sites are given in parentheses.

[b] Rounded to total 100 percent.

Source: DOE79.

Table 1.5 Predicted methods of mining ore reserves

Mining Method	MT of U_3O_8	Percent
Underground Mining	547,000	66
Surface Mining	260,400	31
Other: In situ leaching, heap leaching, & mine water	27,200	3
Total	834,600	

Note.--These are reserves of the $50 per pound U_3O_8 or less cost category.

Source: DOE79.

Although some mines produced much more than others, one can compute a rough estimate of the average capacity of underground and surface mines as follows. The average grades of ore removed from underground and surface mines in 1978 were reported to be 0.155 percent and 0.120 percent U_3O_8, respectively (DOE79). Dividing the annual U_3O_8 productions (Table 1.4) of the two mining methods by their respective grades indicates that the 305 underground mines accounted for 5,387,100 MT of ore while 7,258,300 MT were removed from the 63 surface mines. Hence, the average ore production capacities of underground and surface mines are about 1.8×10^4 MT and 1.2×10^5 MT, respectively. From this assessment, the average ore capacity of a surface mine is about seven times that of an underground mine.

The trend during the past few years of an increasing percentage of U_3O_8 being mined underground will continue, because shallow deposits of high grade ore have tended to be surface mined first. Table 1.5, which displays the distribution of $50 reserves by mining method, shows the continued increase in the proportion of U_3O_8 mined by the underground method. By the Department of Energy predictions, future production from underground mines will more than double that from surface mines. The NRC predicts U_3O_8 production by in situ leaching to peak in 1990 at about 4000 MT/yr and total 76,000 MT by the year 2000 (NRC79). If this prediction is realized, this resource will undoubtedly draw from those assigned to underground and surface mining in Table 1.5. The production of U_3O_8 by heap leaching and mine water extraction is predicted to be relatively small.

It is very difficult to predict how much U_3O_8 will be produced as a by-product from other mineral mines, but by-product production should increase total U_3O_8 production during the next 20 years. Approximately 180 MT of U_3O_8 are currently recovered each year from the phosphoric acid production at wet phosphate plants. The NRC predicts that this will increase to 1800 MT/yr by 1985 and possibly to 7000 MT/yr by the year 2000 (NRC79). If planned leaching facilities are actually built at copper waste dumps at Yerington, Nevada; Butte, Montana; and Twin Buttes, Arizona, to supplement the operating facility at Bingham Canyon, Utah, recovery of U_3O_8 from these operations could reach 900 MT/yr (NRC78). Hence, by-product U_3O_8 production could conceivably account for about eight percent of the required annual U_3O_8 production by the year 2000.

Although the depth of the ore deposit is the fundamental consideration in selecting the mining method to be applied to a particular ore body, the size and grade of the deposit are also important factors. The shape of the deposit, the overburden rock strength, environmental considerations, and other factors may also influence the selection. Surface mining is generally used for relatively shallow deposits; rarely for those below 400 feet (St78). However, under some conditions, it may be cheaper to mine a small, shallow, high-grade ore deposit by underground methods; whereas, a larger low-grade deposit at a greater depth may be cheaper to mine by surface mining. Because productivity is greater by surface mining, it is generally preferred when conditions are favorable. Other factors must be examined when considering the use of in situ leaching (see Section 1.3.4).

1.3.2 Surface Mining

The use of surface (open pit) mining methods is most prevalent in the Gas Hills Region and the Shirley and Powder River Basins in Wyoming, the Laguna District of New Mexico, the coastal plains in south Texas, and some areas of Colorado and Utah (St78). Fig. 1.4 illustrates a typical open pit mine.

In surface mining, an open pit is dug to expose the uranium deposits. After the topsoil is removed and stockpiled nearby, the overburden is removed by the method best suited to the nature of the rock. If the rock is easily crumbled, it is removed by tractor-mounted ripper bars, bulldozers, shovels, or pushload scrapers; if it is not, blasting and drilling are required. The broken rock is then trucked to a nearby waste dump. Occasionally, dikes and ditches are constructed around these waste piles to collect runoff and divert it to sedimentation ponds. Overall, an area of a hundred or more acres may be covered by stored overburden wastes (AEC74).

As mining progresses, the overburden is used as it is removed to backfill mined out areas of the pit. When an area is completely backfilled, it is graded to conform to the surrounding topography and to restore the natural drainage patterns. The area is then covered with topsoil and seeded to blend with the natural terrain. Most of the older surface mines were not backfilled (see Section 3.7.1), and neither are many of the currently active surface mines.

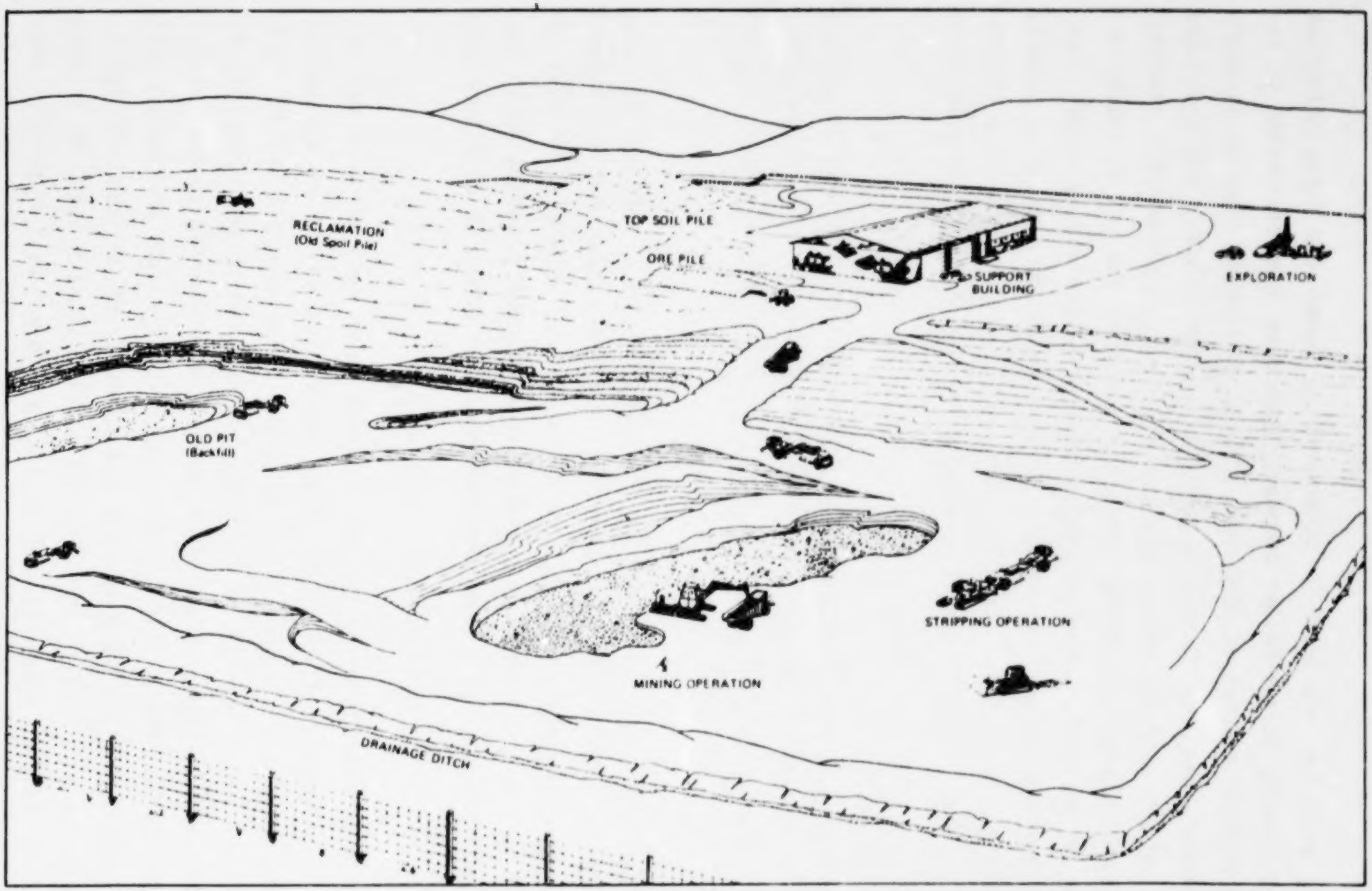


Figure 1.4 Artist's conception of open pit mining operation and support facilities (TVA78a)

Contact with the ore zone is determined by gamma and x-ray detection instruments, usually Geiger-Muller counters that have been calibrated to indicate the uranium content of the rock. Since uranium is usually in sandstone formations, the ore is easily removed. Large backhoes and front-end loaders are often used to remove ore that has been loosened with tractor-mounted rippers. Large ore trucks carry the ore from the pit to stockpiles at the mine or the mill. Uranium ore is usually stockpiled by grade, e.g., high-grade, average-grade, low-grade. Sub-ore grade rock is also usually stored separately in piles. This is rock that contains U_3O_8 at a concentration below what the mill will now accept, but which might later be worth recovering.

Drifts (small tunnels) are sometimes driven into the pit wall to recover small, narrow ore pods. The drifts are generally short, sometimes less than 30 meters. The mining techniques in these drifts are like those used in underground mining (see Section 1.3.3).

Surface mining requires a network of roads from and around the pit area and to the mill. Heavy vehicles operating on these roads, and the digging itself, produce a certain amount of rock and ore dust. However, the dust can be kept down by routinely sprinkling the roads with water or using other dust suppressants. Treated water from the sedimentation ponds is sometimes used for this purpose.

The ratio of overburden to the ore produced in an open pit mine can vary from 10:1 to as high as 80:1 (St78). One source has estimated the average ratio as 30:1 (Le77). A recent study of eight large open pit uranium mines reported a ratio of 77 (± 36) to 1 (Ni79). Since the latter study did not consider the many smaller surface mines where the overburden to ore ratio is likely to be smaller than 77 to 1, this report will assume an average ratio of 50:1. Considering that the average ore capacity of an open pit mine is approximately 1.2×10^5 MT/yr (see Section 1.3.1), about 6×10^6 MT of overburden must be removed annually and initially stored on the surface until reclamation procedures can be initiated.

Since most uranium deposits lie below the water table, groundwater must be prevented from flooding the mining area. One method is to surround the pit with several large capacity wells to lower the water table near the pit. This water is discharged directly into the natural surface drainage system, in accordance with the National Pollutant Discharge Elimination System (NPDES) discharge permit issued to the mining company. Water that does collect in the pit (mine sump water) is pumped to a sedimentation pond for solids removal and, if necessary, for subsequent treatment prior to discharge into the natural drainage system. Another mine dewatering procedure often used consists of ditches dug along the interior perimeter of the pit floor to channel the water to sumps located at the lowest levels of the pit floor. Water that collects in the sumps is pumped to one or more sedimentation basins for solids removal, possible treatment, and final discharge into the existing natural drainage system in accordance with water quality standards specified in the NPDES permit. The rates at which mines are dewatered range from 0.28 m^3/min to 288 m^3/min (AEC74, TVA78a, NRC77a, NRC77b).

Barium chloride, to coprecipitate radium, and a flocculent (an agent causing aggregate formation) to remove other contaminants are usually added to pond water before it is discharged. Water with a high concentration of dissolved uranium is often run through ion exchange columns, and the resin regenerant solution containing the uranium is sent to the mill for processing. The precipitated sludge that collects on the pond bottom consists primarily of ferric and calcium hydroxides, calcium sulfate, and barium sulfate with coprecipitated radium. At some sites, this precipitated sludge is transferred to the mill tailings pond at the end of the mining operation.

A small amount of uncontrolled seepage may occur through the bottom of sedimentation ponds and, depending upon soil permeability and direction of flow, may enter the water table. For example, the seepage rate through the bottoms of two settling ponds totaling 4.9 hectares at one site was less than 0.57 m^3/min (NRC77a). In addition, seepage can be reduced by lining the ponds (well-compacted bentonite clay is sometimes used for this purpose) and by the sludge that accumulates on the pond bottom.

During active surface mining operations, a total of several thousand hectares of land area will be disturbed (St78, Th79). When all uranium has been mined and the operation is completed, a pit remains. The walls of the pit may be contoured and allowed to fill with water, creating a small manmade lake.

1.3.3 Underground Mining

Underground mining is much less disruptive to the surface terrain than open pit mining. The surface affected generally involves less than 41 hectares, but the mine may extend laterally underground for more than a mile and at several depths. Figure 1.5 illustrates a typical large, contemporary underground mine.

In underground mining, access to the ore body is gained through one or more vertical shafts, generally sunk to a slightly greater depth than the ore body, or through inclines, declines, or adits, all cut through waste rock. The waste rock is removed to a spoils area that may be, but usually is not, surrounded by a ditch to contain runoff, as discussed above.

The sizes of the accesses vary considerably. The vertical haulage shaft may vary from less than 8 feet in diameter, sufficient to accommodate one small ore skip (a large bucket), to a diameter of 20 feet, which will accommodate dual ore skips as well as a man and material skip. In some cases, the near horizontal accesses are sufficiently large to allow passage of large diesel-powered vehicles.

Underground mines are developed in a way that minimizes the removal of waste rock, resulting in much smaller spoil storage piles than those at surface mines. It is estimated that the ore to waste rock ratio generally ranges from 20:1 to 1:1 (ACE74, Th79). At seven presently active mines, the ore to waste rock ratio ranges from 1.5:1 to 16:1 with an average ratio of 9.1:1 (Ja80). Using the average ratio and the average annual ore capacity of an underground mine (see Section 1.3.1), each year the average underground mine will produce about 2.0×10^3 MT of waste rock that is removed and stored on the surface. Initially all waste rock is transported to the surface, but,

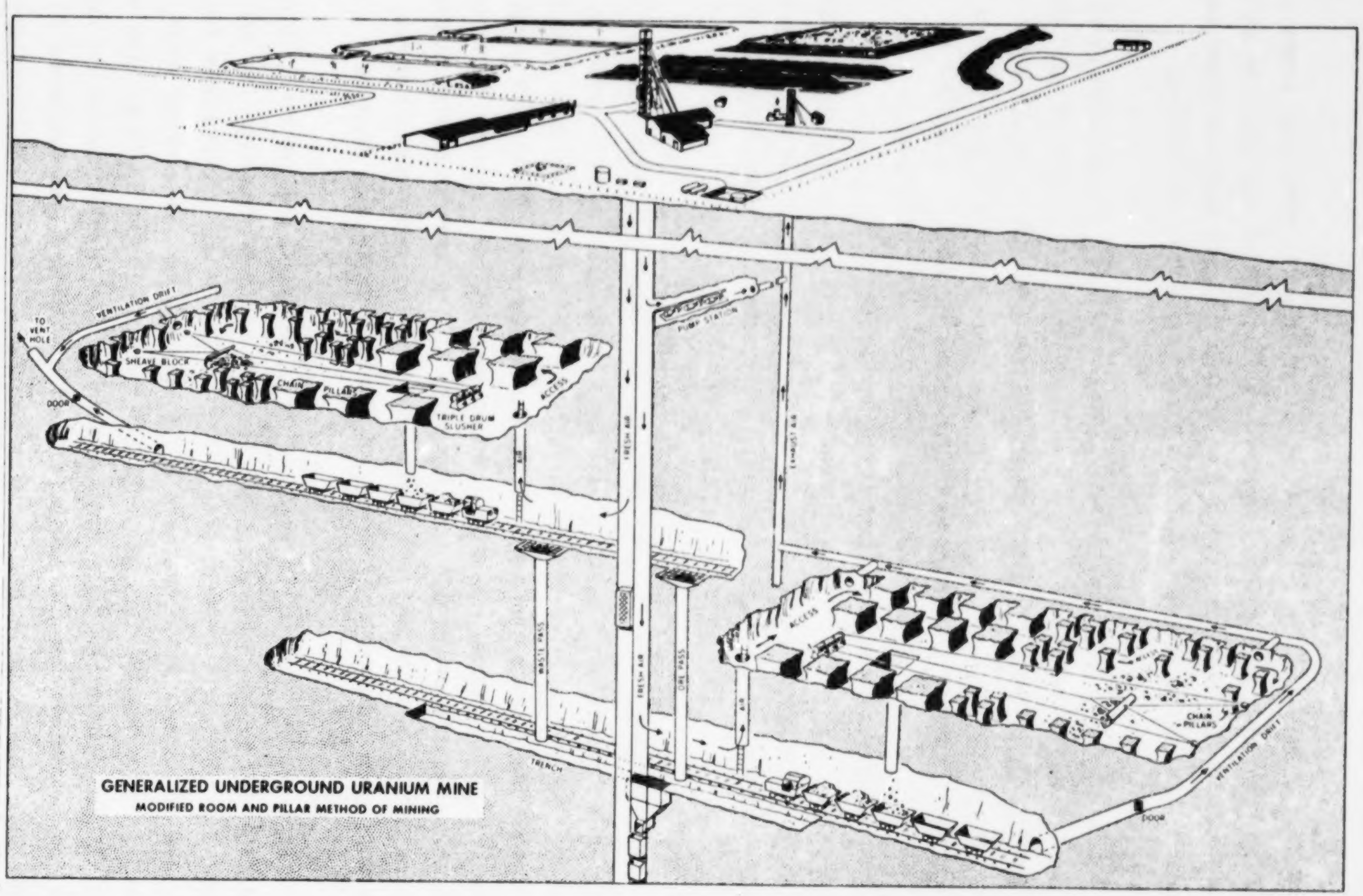


Figure 1.5 Generalized underground mine showing modified room and pillar method of mining (TVA78b)

as mining progresses, it is sometimes transported to mined-out areas of the mine and retained beneath the surface. This practice may diminish as lower grade sub-ore becomes more economical to mill. Since waste rock may contain sub-ore, some waste rock will likely be kept available for milling.

Ore deposits, outlined by development drilling, are followed as closely as possible. When ore lies in narrow, long deposits, drifts are cut through the ore body and raises or stopes are driven from the drift to reach small ore pods. Crosscuts are driven from the haulage drifts when necessary to reach nearby deposits. Large area deposits are commonly mined by the "room and pillar" method. This involves mining out blocks of ore while leaving adjacent pillars of ore or waste as support for the roof. The size of the rooms depends on the roof condition. The roof is usually strengthened by bolts, wire mesh, timbersets, and steel arches. When an area is completely mined, the ore pillars are removed in a systematic sequence that allows safe retreat.

Ore is usually broken by drilling and blasting. The broken ore is removed and transferred to mine rail cars. The ore is then carried by rail cars or wheeled vehicles either directly to the surface or to a skip at the bottom of the haulage shaft and lifted to the surface. Haulage in large area mines is often accomplished by large diesel-powered loaders, haulers, and trucks. When the ore is sufficiently soft, it may be removed with continuous mining machines instead of drilling and blasting techniques; however, most ore bodies are too small and irregular to mine economically this way.

Ventilating systems are required in underground uranium mines to remove blasting fumes and radon-222 (Rn-222) that emanates from the ore and mine water and to control temperature. Fresh air is usually forced down the main haulage shaft and along the main haulage drifts to the working areas. The mine air is exhausted through ventilation shafts to the surface. The ventilation air is diverted from inactive areas of the mine to reduce air contamination. Inactive areas are usually sealed with airtight bulkheads to prevent radon gas in those areas from circulating. The ventilation rate should be sufficient to maintain the radon daughter concentration of the mine air at, or below, levels that meet federal and state occupational exposure standards. The rate will vary depending upon mine size (volume), grade of exposed ore, size of the active working areas, rock characteristics (diffusion rate of Rn-222), effectiveness of bulkhead partitions, atmospheric pressure, and other factors. Ventilation rates in active mines vary from a few hundred m^3/min to over a hundred thousand m^3/min. For example, the ventilation rates for seven uranium mines in the Grants, New Mexico area ranged from 4.4×10^3 to 1.1×10^4 m^3/min, with an average of 7.4×10^3 m^3/min (Ja79).

Because ore bodies often lie in or beneath major aquifers, dewatering operations similar to those practiced in surface mining are required. These operations commence during the initial shaft-sinking process and may continue throughout the working life of the mine. Water is pumped from wells that are driven into the water-bearing strata near the mining operation and discharged either directly into the natural surface drainage system, in accordance with an NPDES permit, or to settling ponds. Water that collects in the mine is diverted to sumps and pumped to a settling pond. The impounded mine water is treated similarly to that described above at surface mines (see Section 1.3.2). The discharge of water from these ponds is in accordance with water quality standards specified in the NPDES permit. (Note.--About one-half of the active New Mexico mine discharges have NPDES permits that are presently under adjudication and, therefore, are not necessarily in accord with discharge limits [Pe79a].)

1.3.4 In Situ Leaching

In situ leaching has less adverse impact on the environment than conventional uranium mining and milling methods. It also may permit economical recovery of currently unrecoverable low-grade uranium deposits (NRC78). Though in situ leach mining currently produces only a small amount of the annual U.S. output of U_3O_8, variations of this technique are being widely tested for uranium extraction and have potential for becoming commercially significant (La78, NRC78, TVA78b, Ka78). Table 1.6 lists in situ leaching operations for uranium as of January 1, 1978. The operations are concentrated on the coastal plain of southwest Texas and in the Wyoming basin regions. Most commercial sized operations are in southern Texas, where recent expansion is expected to increase the production of U_3O_8 by this technique to about 900 MT annually (TVA78b). Two Texas sites alone, Bruni and Lamprecht, are expected to produce annually 110 and 230 MT of U_3O_8, respectively (Wy77). A number of projects are currently testing the effectiveness of the in situ leaching technique. Though these studies usually last about 18 months to 3 years, some feasibility tests require up to 6 years before expanding to full or commercial scale operations (La78). Excellent reviews of this mining method are available (La78, Ka78).

Uranium extraction by in situ leaching probably will not be restricted to one or two geographical areas. Uranium deposits potentially suitable for mining by this method are prevalent in almost all of the established uranium mining areas in the United States. Uranium deposits are potential candidates for in situ mining if they meet the following criteria: (1) the ore deposit is located in a zone saturated with water; (2) the ore deposit lies above and preferably between geological layers impervious to water; (3) the deposit is adequately permeable to water; and (4) the uranium in the ore deposit is in a leachable state. Colorado and New Mexico already have in situ leaching activities at the pilot scale, and the mining industry has inquired about additional pilot-scale research and development sites in South Dakota, Arizona, Utah, and Montana (La78).

In the in situ leaching method, a leaching solution (lixiviant) is injected through wells into the uranium-bearing ore body. It forms chemical complexes with the uranium, which dissolves in the solution. Production wells bring the uranium-bearing solution to the surface where the uranium is extracted. The barren lixiviant can then be reconstituted and reused. To control groundwater flow, the production (pumped) well operates as a sump or pressure sink in the formation, which produces a flow of groundwater and lixiviant from the injection wells to the production well. Also, some of the barren lixiviant is not reinjected. This reduces the water level in the well field, allowing groundwater to migrate into the mining zone. This inflow prevents the flow of the lixiviant away from the field area.

Lixiviants for in situ mining contain salts of anions (negatively charged chemical groups), such as sulfate, carbonate, bicarbonate, and ammonium, that form stable aqueous complexes with hexavalent (positively

Table 1.6 Summary of current in situ leaching operations as of January 1, 1978

Name	Location	Well[a] Pattern	Scale of[b] Operation	Flow Rate[c] (m^3/min)
Sundance Project	Crook County, WY	5-SP	RD-PS	ND[d]
Red Desert Site I	Sweetwater County, WY	5-SP	RD-I	ND
Red Desert Site II	Sweetwater County, WY	5-SP	RD-PS	ND
Charley Site	Johnson County, WY	5-SP	RD	ND
Highland Site	Converse County, WY	7-SP	RD-C	4.54
Double Eagle Site	Carbon County, WY	5-SP	RD	ND
North Rolling Pin Site	Campbell County, WY	5-SP	RD-I	ND
Collins Draw Site II	Campbell County, WY	5-SP	RD	0.38-0.57
Bear Creek Site	Converse County, WY	5-SP	RD-I	ND
Nine Mile Lake Site	Natrona County, WY	5-SP	RD-PS	0.38
Red Desert Site	Sweetwater County, WY	5-SP	PS	0.38
Irigaray Site	Johnson County, WY	ND	C	6.06
Site No. I	McKinley County, NM	4-SP	PS-I	ND
Site No. 2	Sandoval County, NM	4-SP	PS-I	ND
Crownpoint Project	McKinley County, NM	4-SP	RD	ND

Table 1.6 Summary of current in situ leaching operations as of January 1, 1978 (continued)

Name	Location	Well[a] Pattern	Scale of[b] Operation	Flow Rate[c] (m^3/min)
Grover Site	Weld County, CO	5-SP	PS-C	0.76
Palangana Dome Site	Duval County, TX	ND[d]	C	ND
O'Hern Site	Duval County, TX	ND	C	ND
Bruni Site	Duval County, TX	ND	C	ND
Lamprecht Site	Bee County, TX	ND	C	0.76
Zamzow Site	Live Oak County, TX	ND	C	ND
Boots/Brown Site	Live Oak County, TX	ND	C	ND
Clay West Site	Live Oak County, TX	ND	C	ND
Burns Ranch Site	Live Oak County, TX	ND	C	ND
Moser Site	Live Oak County, TX	ND	C	ND

(a)Well pattern: 5-SP indicates one or more 5-spot pattern(s), etc. See Figure 1.6.

(b)Given are past or present operations - planned future operations: RD-research and development, PS - pilot scale, C - commercial scale, I - presently inactive.

(c)Flow rate of leachate to processing plant in m^3/min.

(d)No data.

Source: La78; Du79.

charged in the +6 state) uranium. An oxidant, such as air, oxygen, hydrogen peroxide, sodium chlorate, sodium hypochlorite, or potassium permanganate, is added to oxidize the uranium to the hexavalent state. For example:

$$UO_2 + H_2O_2 + 2HCO^-{}_3 \rightarrow UO_2(CO_3)_2^{=} + 2H_2O$$

Unfortunately, there is no lixiviant specific for uranium. Consequently, other minerals commonly associated with uranium deposits, such as iron, selenium, vanadium, molybdenum, and arsenic, may also be dissolved. This tends to contaminate the leach solution and deplete the lixiviant. Lixiviant agents and their concentrations are selected to maximize uranium recovery and minimize undesirable secondary reactions. Acidic solutions (pH 2) are avoided because they are less selective. Neutral or basic lixiviants (pH 6-10), such as ammonium or sodium carbonate or bicarbonate, are often used.

Many variables affect the accumulation of trace elements in leaching solutions, particularly the chemical and physical nature of the host formation. Table 1.7 illustrates relative contaminant levels in the two lixiviant types in a laboratory experiment. Except for Ra-226, significantly greater trace element concentrations occur in the acid lixiviant; the total dissolved solids is about eight times higher than in the alkaline solution. Hence, it would be necessary to bleed much larger volumes of acidic lixiviant from the system prior to reinjection in order to maintain acceptable levels of these undesirable constituents. Large volumes of liquid wastes containing higher toxic metal concentrations are generally produced when acidic lixiviants are employed. Also, because calcium minerals are abundant in geologic strata and carbonate minerals are highly soluble in acid solutions, particularly calcium carbonate, large amounts of calcium accumulate in recirculated acid lixiviant, and they must be removed by a purification process prior to reinjection. However, acid lixiviants leach more rapidly than alkaline ones, yield higher uranium recoveries -- about 90 percent with sulfuric acid compared to 60 to 70 percent with a bicarbonate solution -- and generally extract less radium (Wy77).

The number of wells, their spacing, and their pattern depend upon the size and hydrologic characteristics of the formation. Figure 1.6 shows diagrams of some common well patterns. Several hundred injection wells with several recovery wells may be employed. Well spacing may vary from 10 to 60 m. In addition, a number of monitoring wells are driven a short distance from the well field to detect any excursion of lixiviant from the leach field. A commercial-size operation may require a well field area of 20 hectares or more (TVA78b).

The pregnant (containing uranium) leachate from the production wells is filtered through a sand filter to remove suspended particulates, then passed through a surge tank (storage reservoir) to ion-exchange resin beds that selectively remove the uranium complex. The uranium is washed from the resin beds, precipitated, filtered, dried (at most sites), and packaged.

Some processes of solution mining produce liquid and solid wastes. The volume of liquid wastes produced is much smaller, per weight of U_3O_8 produced, than that from the dewatering activities of conventional mining methods. There is also no waste rock. Residues obtained from drilling are a solid waste. Those that traverse the ore zone will contain some uranium ore. If calcium or sulfate control of the lixiviant is necessary, additional solid wastes are impounded in waste ponds under a liquid seal to minimize atmospheric dispersion. Precipitation compounds will also be produced as evaporation concentrates the impounded waste solutions.

Liquid waste streams include lixiviant, filter and resin washes, resin eluant bleed, and water used in cleaning the injection wells. The total production rate of these waste streams may vary between 0.19 to 0.38 m^3/min (Ka78, Wy77, TVA78b). At most sites, all liquid wastes flow to waste ponds and evaporate. Pond size depends on the flow rate of the wastes and the evaporation rate. The pond bottoms are usually lined with clay, asphalt, or a continuous plastic sheet to minimize the seepage rate, although some seepage may inadvertently occur. Deep-well injection is also used, principally in Texas, to dispose of liquid wastes from in situ leaching (Du79).

Table 1.7 Trace metal concentrations of recirculated acid and alkaline lixiviants

	Concentrations, mg/ℓ	
Trace Metal	Acidic[a]	Alkaline[b]
Arsenic	<0.05	<0.05
Chromium	0.15	0.07
Cobalt	0.2	NR[c]
Copper	1.0	0.04
Iron	25.4	0.6
Lead	0.7	0.2
Manganese	1.2	NR
Molybdenum	NR	0.9
Nickel	0.6	0.06
Selenium	NR	1.6
Strontium	3.7	1.5
Vanadium	1.0	NR
Zinc	4.3	0.1
Zirconium	3.3	0.9
Radium-226[d]	390	1750
TDS[e]	7.8	1.0

(a) Composition - 5 g/ℓ H_2SO_4 and 0.1 g/ℓ $NaClO_3$.

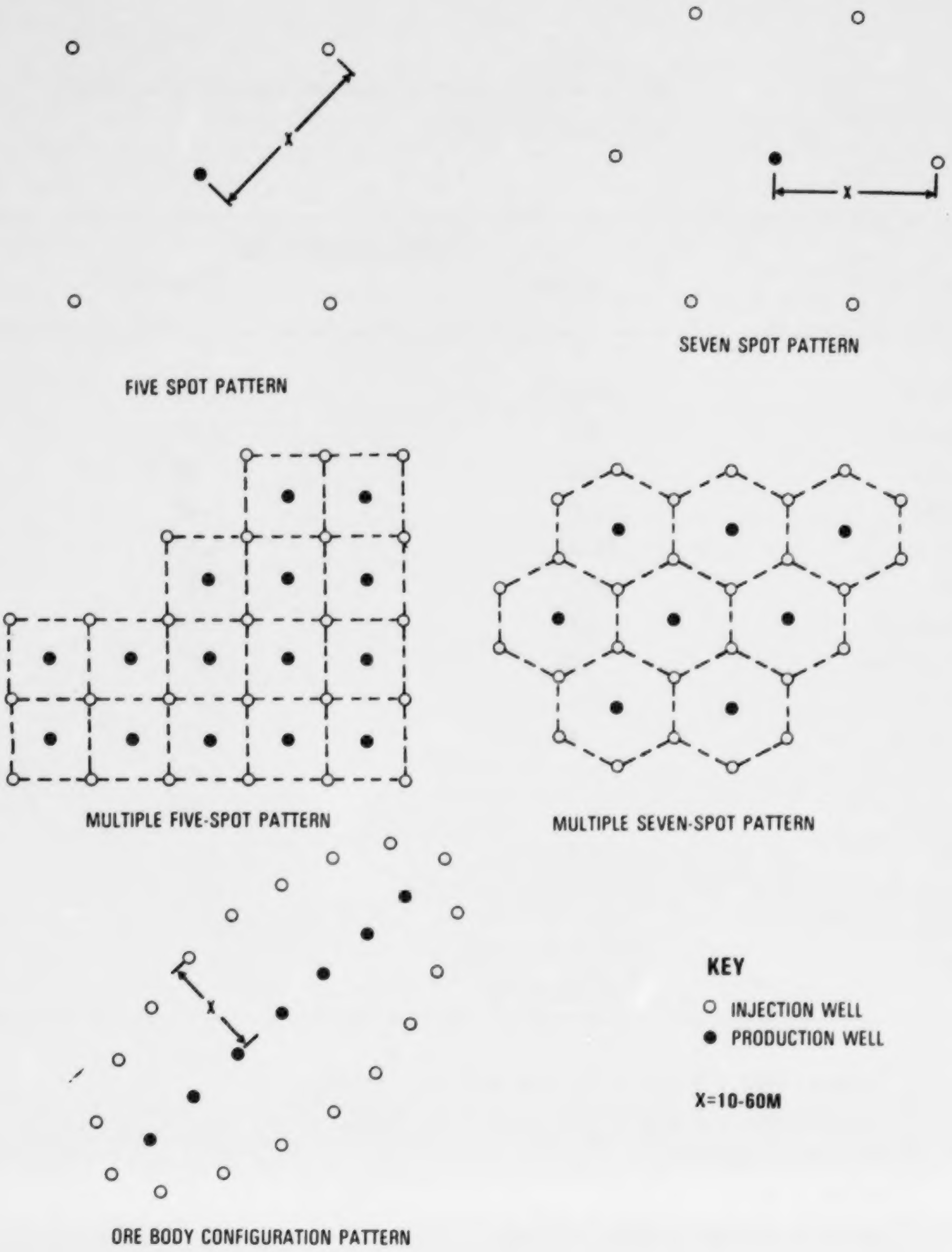
(b) Composition - 8 g/ℓ NH_4HCO_3 and 1 g/ℓ H_2O_2.

(c) NR - Not Reported.

(d) Units - pCi/ℓ.

(e) Total dissolved solids in grams.

Source: Ka78.


FIVE SPOT PATTERN

SEVEN SPOT PATTERN

MULTIPLE FIVE-SPOT PATTERN

MULTIPLE SEVEN-SPOT PATTERN

ORE BODY CONFIGURATION PATTERN

Figure 1.6 Diagrams of some common injection-recovery well patterns used in uranium in situ leach mining

Atmospheric emissions from in situ leaching include Rn-222 that is vented mainly from the pregnant lixiviant surge tanks and particulate matter that may escape from the scrubbers (exhaust filters) of the yellowcake (uranium product) drying and packaging units. Radon-222 emanation from the waste ponds is probably negligible, since the sediment which contains the radium remains submerged and little Radon-222 will diffuse through the water and escape to the atmosphere.

When the mining operation is completed, the water volume of the leach zone will be restored to limits set by regulatory agencies. The primary method of aquifer restoration is flushing the zone with groundwater by pumping from the production wells and/or the injection wells. This process may produce up to 1.70 m^3/min additional liquid wastes that contain a high concentration of dissolved solids (NRC78). This contaminated water passes through an ion-exchange unit and then discharges to the waste ponds. The barren effluent can be further treated by desalination and reinjected to the formation. Also, a barium chloride solution can be injected into the leach zone to coprecipitate radium from the aquifer water prior to the groundwater sweeps.

1.3.5 Other Mining Methods

1.3.5.1 Heap Leaching

Ore of a grade too low to be economically extracted in the mill operation is sometimes treated by heap leaching. In this process, the low grade ore is placed in large, rectangular, open-air piles on a specially prepared pad. To construct the heap-leach pad, the topsoil is first removed and the cleared area graded to a 2 percent to 3 percent slope. The graded area is then covered by a plastic sheet. Perforated plastic pipe is placed on the plastic parallel to the slope and covered with approximately 30 cm of clean, coarse gravel. A collecting trough is formed at the base of the slope and a berm surrounds the pile area.

When the ore is dumped on the pad, large solution reservoirs are formed on the top. Acidic or alkaline mine water is pumped into the reservoirs and percolates through the pile. The percolated water is collected in the trough and recirculated until the concentration of uranium in solution is sufficient to be economically extracted. The leaching process may require up to six months to recover approximately 80 percent of the uranium in the ore (NRC77a). The heap-leach pile at one mine contained an annual accumulation of approximately 360,000 MT of low-grade ore (NRC77a). The pile measured 300 m x 90 m x 7.6 m with solution reservoirs of 22 m x 90 m x 1.5 m.

After leaching operations are completed, the leached pile is neutralized with lime to a pH of about 7. The site is then contoured to blend with the surrounding terrain, covered with layers of subsoil and topsoil, and seeded to control wind and water erosion.

Because necessary information is unavailable and the contribution of heap leaching to the total uranium production is very minor and not expected to become significant (NRC79), an assessment of the environmental impact of heap-leached piles has not been conducted. However, the NRC has recently concluded that, although the hazard of tailings produced by heap leaching will be much less than the hazard of tailings at conventional uranium mills, the same tailings management and disposal criteria should possibly apply (NRC79).

1.3.5.2 Mine Water Recirculation

At several sites mine water is recirculated to leach "worked-out areas" of underground mines (Pe79b). In the early uranium mining years, ore with less than about 0.15 percent U_3O_8 was not mined. This grade is relatively high compared to present day markets. Consequently, significant quantities of uranium remain in these abandoned areas. Because the roofs of these areas collapsed during the initial mining retreat, this ore is difficult to retrieve by conventional methods. To recover a portion of this uranium, holes are drilled to the top of the collapsed zone and mine water is sprayed from these holes onto the shattered ore. Water for leaching may be sprayed from the mine floor if the abandoned area is accessible to the workers (Pe79a). The oxidized uranium (uranyl ion) is leached by the slightly alkaline mine water, which flows to collection sumps. The enriched water is pumped to a resin ion-exchange unit to extract the uranium, and then it is recycled.

After the available oxidized uranium has been leached, the process is discontinued for a few weeks to allow more uranium to oxidize. Mine water is then circulated again through the ore.

This process increases the recovery of uranium with minor effort and expense, but it contributes little to the total domestic uranium production (NRC79). In addition, the quality of the stored mine water used will be enhanced after passing through the resin ion-exchange unit. Hence, mine water recirculation has little impact on the environment. It was not assessed in this study.

1.3.5.3. Borehole Slurry Mining

Hydraulic borehole slurry mining is a recently proposed technique for extracting uranium ore (Ka78, St78). As the name suggests, this method uses pressurized water to loosen and combine with ore-bearing material to form a watery mixture known as 'slurry' that is transported from the borehole and then conventionally milled. This method could be applied to sandstone deposits at depths of 30 m to about 100 m. By present estimates, yellowcake from ore containing 0.06 percent U_3O_8 and mined at a 60 m depth by this method would cost $42 per pound (Ka78). This method presently is not as economical as the more conventional methods of uranium mining.

The process consists of drilling a 45-cm diameter hole to approximately 2 m below the uranium-bearing strata. A cutting jet assembly is positioned in the hole at the end of a rigid service column containing conduits for the pressurized water and slurry transport. The slurry pump is placed at the bottom of the hole. The underground mining operation is started with the jet set at the lowest position. The rotating jet cuts material through an arc of somewhat less than 360^0 for a distance of up to 25 m, depending upon the design of the jet system. The segment of unmined ore acts to support the overlying strata. After the material is removed as a slurry, the jet is raised to the next level of ore and the process is repeated. After milling, the decanted water from the slurry is recycled for slurrying more ore. The tailings from the milling operation are used to backfill the borehole cavities and minimize subsidence.

A 15 m to 25 m radius borehole can be mined in an 8- to 24- hour period (Ka78). Large ore bodies might be mined by drilling, slurrying, processing, and backfilling in a systematic pattern that leaves ore in between boreholes for support. These areas could be mined in a second phase after the original boreholes are backfilled.

Borehole mining for uranium is currently only a proposed method with no pilot or commercial scale units in operation. Thus, the possible environmental impact from this process was not assessed in this study.

1.3.5.4. Uranium as a By-Product

The recovery of uranium as a by-product from other mineral mining and milling operations was discussed briefly in Section 1.3.1. Since recovery is basically from the milling operation, any environmental problem that might exist is associated with milling rather than mining. Therefore, it was not assessed in this study.

1.4 Current Applicable Standards and Regulations

1.4.1 Federal Regulations

Health, safety, and environmental hazards associated with uranium mining are regulated by Federal and State laws. This review focuses on laws and regulations applicable to mine operations. Nuclear Regulatory Commission regulations for milling operations apply to in situ leach extraction and are therefore included. Some laws and regulations on exploration rights also cover the environmental impact of mining operations and wastes.

Prior to the National Environmental Policy Act (NEPA) of 1969, there were few regulations protecting the environment of lands not controlled or owned by the Federal Government. Even with NEPA, much Federal authority on environmental problems was unused until recently. This Act established a national policy concerning the environment. Section 102(2) (C) states that every agency of the Federal Government must "include in every recommendation or report on proposals for legislation and other major Federal actions significantly affecting the quality of the human environment, a detailed statement" of the environmental impact of such an action. Major Federal actions

"... includes actions with effects that may be major and which are potentially subject to Federal control and responsibility ... actions include new and continuing activities, including projects and programs entirely or partly financed, assisted, conducted, regulated, or approved by Federal agencies; new or revised agency rules, regulations, plans, policies, or procedures; and legislative proposals (Sections 1506.8, 1508.17) . . . . Approval of specific projects, such as construction or management activities located in a defined geographic area. Projects include actions approved by permit or other regulatory decision as well as federal and federally-assisted activities"(40 CFR 1500).

1.4.1.1 Federal Laws, Regulations, and Guides for Protection of Health and Environment

Table 1.8 provides an overview of federal laws and regulations for the protection of health or environment and the administering agencies. Federal agency responsibilities for water use, conservation laws, and exploration and mining rights are indicated in columns 1-4. Laws and regulations for environmental quality and health and safety are indicated in columns 5-10. See Appendix A for an itemized list of the laws and regulations shown generally in Table 1.8.

1.4.1.1.1 Air Quality

Regulations on air quality have been promulgated pursuant to the Clean Air Act (42 U.S.C. 1857 et seq), which includes the Clean Air Act of 1963 (Public Law 88-206) and amendments by the following: Public Law 89-272, Public Law 89-675, Public Law 90-148, Public Law 91-604, Public Law 92-157, Public Law 93-319, Public Law 95-95, and Public Law 95-190. The Environmental Protection Agency establishes National Ambient Air Quality Standards, New Source Performance Standards, and National Emissions Standards for Hazardous Air Pollutants under the Clean Air Act (CAA). Primary standards are set to protect public health and secondary standards are set to protect public welfare from known or anticipated adverse effects.

National Ambient Air Quality Standards (NAAQS) have been established for seven pollutants in 40 CRF 50. The Administrator of EPA is authorized to set emission standards for hazardous air pollutants for which no ambient air quality standard is applicable. Asbestos, beryllium, mercury, and vinyl

Table 1.8 Federal laws, regulations, and guides for uranium mining

Federal Agency	General		Mining							
			Permits		Environmental Quality					Health and Safety
						Water			Land	
	Water Use	Conservation-Preservation Statutes	Exploration Rights	Mining Rights	Air	Surf	UG	Solids	Reclam	
Dept. of Int.	X	X	X	X					X	
BIA[(a)]			X	X					X	
BLM[(a)]		X	X	X					X	
USGS[(a)]			X						X	
Dept. of Energy		X	X	X					X	
Dept. of Agr.	X	X							X	
USFS[(a)]			X	X					X	
EPA	X	X				X		X	X	X
AIR-OAQPS[(a)]					X					
Water										
Surface OWPS[(a)]						X				
Ground OSW[(a)]							X			
Land-OSW[(a)]						X	X	X	X	
Radiation-ORP[(a)]					X	X	X	X	X	X
U.S. Army										
Corps of Engrs.	X	X				X			X	
Dept. of Labor		X								
MSHA[(a)]										X
OSHA[(a)]										X
Nuclear Reg. Comm.[(b)]		X		X		X		X	X	X

[(a)] BIA-Bureau of Indian Affairs
BLM-Bureau of Land Management
USGS-United States Geological Survey
USFS-United States Forest Service
OAQPS-Office of Air Quality, Planning and Standards
OWPS-Office of Water Planning and Standards
OSW-Office of Solid Waste
ORP-Office of Radiation Programs
MSHA-Mining Safety and Health Administration
OSHA-Occupational Safety and Health Administration

[(b)] Nuclear Regulatory Commission (NRC) regulations and guides for milling do apply to in situ extraction or mining but not conventional surface or underground mining where NRC has no authority.

chloride emission standards are in subparts, B, C, E, and F of 40 CFR 61, respectively. Section 122 of the CAA directed the Administrator to determine whether emissions of radioactive pollutants, cadmium, arsenic, and polycyclic organic matter (such as benzene) into ambient air will cause or contribute to air pollution and endanger public health. If they do, EPA must propose emission standards for them within 180 days after that decision. The EPA has listed radionuclides as "hazardous pollutants" under Section 112 of the Clean Air Act in December 1979 (44FR76738, December 27, 1979). To date, no standards for radionuclide emissions in air have been promulgated.

The particulate concentration values of the NAAQS apply to mining operations. Emissions (including dust) must be controlled to meet the standards. Dust from mining operations was excluded from any air quality impact assessment for prevention of significant air quality deterioration (PSD) (see 43 F.R. 26395). However, as a result of the court decision in Alabama Power Company v. Costle, 13 ERC 1225, EPA has proposed amendment of PSD regulations (44 F.R. 51924, September 5, 1979).

The emission of radioactive substances or gases from gaseous release is controlled by NRC regulations 10 CFR Parts 20 and 40 for uranium milling and in situ leaching. The NRC does not have this authority over mining. There are no Federal regulations for radioactive pollution of air from mining at this time. However, MSHA enforces standards for radioactivity in air inside mines (30 CFR 57.5-37 through 57.5-42). Health and Safety standards of MSHA for Metal and Nonmetallic Mine Safety are given in 30 CFR Parts 55, 57, and 58.

1.4.1.1.2 Water Quality

Standards for water quality are promulgated by EPA under the Federal Water Pollution Control Act (FWPCA) of 1948 (as amended) and the Safe Drinking Water Act (SDWA) (as amended). The FWPCA and SDWA regulate surface water quality and groundwater quality, respectively.

The Federal Water Pollution Control Act Amendments of 1972(Public Law 92-500) established that no one has a right, without permit, to discharge pollutants into navigable waters of the nation. The Act provides for the establishment of both water quality standards and effluent limitations. In addition to requiring effluent standards for existing sources, it required EPA to set new source performance standards for uranium mining. The following standards and guidelines apply to uranium mining and milling: Regulations on Policies and Procedures for the National Pollutant Discharge Elimination System (40 CFR 125), Effluent Guidelines - Mining and Processing (40 CFR 116), Effluent Guidelines and Standards for Mining and Processing (40 CFR 436), and Protection of the Environment-Ore Mining and Dressing - Point Source Category (40 CFR Part 440). Table 1.12 lists other pertinent regulations and guides.

The Safe Drinking Water Act primarily protects municipal water systems. Part C of the Act requires that states establish underground waste water injection programs according to EPA regulations. Most mining operations dispose of waste water through surface discharges subject to the NPDES permit program and to the FWPCA. However, if a mine or mill seeks to dispose of polluted water by injection and such injection may endanger public drinking water supplies, then the Safe Drinking Water Act would apply. Finally, EPA will be developing regulations pursuant to Subtitle C of the Resource Conservation and Recovery Act that will provide controls on hazardous uranium mining wastes, including protection of groundwater resources. Section 4004 criteria, promulgated on September 13, 1979, apply to the nonhazardous portion of the wastes.

The NRC's water quality standards for radioactivity in discharges from uranium milling to the environment are in 10 CFR Parts 20 and 40. These would apply to in situ mining licensed by NRC or an agreement state.

1.4.1.1.3 Land Quality

Federal regulations on solid waste disposal and land reclamation specifically for uranium mining wastes are being developed pursuant to the Solid Waste Disposal Act (as amended). The Surface Mining Control and Reclamation Act of 1977 only applies to coal mining. Uranium mining occurs on Federal lands, where the Departments of Interior and Agriculture require reclamation. A large part of the western states is Federally owned land: Arizona (43 percent), California (45 percent), Colorado (36 percent), Idaho (64 percent), Montana (30 percent), Nevada (87 percent), New Mexico (34 percent), Texas (2 percent), Utah (66 percent), Washington (29 percent), and Wyoming (48 percent). State laws and local zoning ordinances may affect waste disposal. Many states authorize counties to regulate land use outside incorporated areas. Likewise, many states allow cities, towns, and villages to enact zoning ordinances for land use within their boundaries. Thus, mining operations in each state are subject to different reclamation requirements, depending upon land ownership and location.

Regulations for hazardous uranium mining wastes have been proposed by the EPA pursuant to Subtitle C of the Solid Waste Disposal Act as substantially amended by the Resources Conservation and Recovery Act of 1976 (Public Law 94-580). These were published in the Federal Register (43 F.R. 58946-59028) on December 18, 1978. Waste rock and overburden from uranium mining are listed as hazardous wastes, because they contain radioactive substances that meet the definition of hazardous wastes given in Section 1004 (5) of the Act. Special waste standards (Part 250.46-4) were proposed for the treatment, storage, and disposal of overburden and waste rock.

1.4.1.2 Federal Mineral Leasing and Location/Patent Laws

Some Federal regulations govern mineral exploration and mining rights. The Mining Law of 1872 (30 USC §§ 21-50) permits persons to enter public lands to discover, locate, and mine valuable minerals. The law has no provisions for facility siting, surface protection, or reclamation. Free use of water and timber for the mining operation and land for a mill site are ancillary rights granted by the law. Most subsequent mineral leasing laws are similar, designed to provide an orderly system for locating, removing, and utilizing valuable mineral deposits on federally owned and controlled lands. Pursuant to Section 603 (C) of the Federal Land Policy and Management Act of 1976, DOI has proposed specific environmental protection regulations (43 CFR 3800) for mining activities in potential or identified wilderness study areas (44 FR 2620).

1.4.1.2.1 Prospecting and Mining Rights

Consideration of environmental impacts may be required before obtaining the right to prospect or explore. Depending upon land category, prospectors may have to assess the environmental impact of mineral exploration before being permitted to explore. Table 1.9 summarizes these requirements. Prospectors on private lands simply must have permission from the owner of record of mineral estate. On the other hand, Tribal and Indian lands, National Forest System lands, and public lands (not public domain) all have specific approval systems that require exploration plans or other appropriate considerations.

Obtaining rights to mine usually involves the same Government agency involved with prospecting rights. Table 1.10 summarizes applicable Federal laws and regulations.

1.4.1.2.2 Mining and Environmental Plans

Before mining begins certain operating or mining and reclamation plans must be submitted and approved. Table 1.11 summarizes these. The requirements parallel those for prospecting and mining rights.

1.4.1.3 Laws Having Potential Applicability

Federal laws require regulation for quality of air, water, and land. In addition, though their direct influence has not been evaluated in this report, federal laws protecting wildlife and cultural resources could affect uranium mining activities.

Water use is also of potential concern in regard to uranium mining. However, except for in situ mining, uranium mining operations have modest needs for water. In fact, most mines typically dispose of significant quantities from necessary dewatering. Appendix B lists federal water programs and rights activities and the lead agencies administering them; and Appendix C lists Congressionally approved compacts that apportion water. These compacts apportion water to the affected states, and each state in turn allocates its share of the water among intrastate users on the basis of its own system of water rights.

Table 1.9 Requirements to obtain rights to prospect or explore by federal, state and private lands

Land Category	Requirements
Federal:	
Tribal............	Prospecting permit issued by BIA with consent of tribe. 25 CFR 171.27a. Technical examination of environmental effects of prospecting by BIA, 25 CFR 177.4. Exploration plan submitted to USGS. Approval of plan by USGS required, 25 CFR 177.6. Enforcement of plan by USGS, 25 CFR 177.10.
Allotted Indian....	No specific provisions for prospecting. Procedure for leasing to prospect is same as for mining. If allotted land has been patented, treat same as private land.
Public Domain......	No restriction on prospecting. Entry under General Mining Law of 1872 (30 USC 22, 43 USC 1744, 43 CFR Part 3810), uranium included, 43 CFR 37461.
Acquired Public....	Prospecting permit from BLM, 43 CFR 3510.0-3 and 3511.2-1. Acquired lands not subject to prospecting permits are listed in 43 CFR 3501.2-1. If acquired land is not under BLM jurisdiction, consent of governmental entity having jurisdiction is required before permit issued by BLM (43 CFR 3501.2-6).
Withdrawn Public...	Public domain land withdrawn for power development is open to entry and location under General Mining Law of 1872, 30 USC 621. Agency having control of withdrawn land reports any objections to mining activity based on land use for which withdrawal was made. If controlling agency recommends stipulations in the permit, they are included (43 CFR 3501.3-1 (a), (c).
Reserved Public....	Some Federal lands are disposed of with minerals reserved to the Government; e.g., see 43 USC 299, 43 CFR 3814.1, 30 USC 50. For these lands, permit issued by BLM requires conformance with law under which reservation was made, 43 CFR 35013-2(2). For lands reserved or segregated for particular purpose, special requirements may be made by BLM for protection and use of land for purpose that it was reserved or segregated. Leases from Dept. of Energy may be possible under 42 USC 2097.

Table 1.9 (Continued)

Land Category	Requirements
National Forest System...	Public domain lands inside National Forest System boundaries are subject to General Mining Law of 1872, with the following conditions: (a) If Dept. of Agriculture requires operations plan, it must be submitted. Dept. of Agriculture approves plan, 36 CFR 252.1; (b) Operations must minimize environmental impact on surface resources in System lands, 36 CFR 252.8; (c) Surface inspection and securing compliance with plan is responsibility of Dept. of Agriculture, 36 CFR 252.7. Acquired National Forest System land same as Acquired Public Land.
State....................	Lease obtained from appropriate State Agency according to state law.
Private.................	Permission given by owner of record of mineral estate.

Source: San Juan Basin Regional Uranium Study, Working Paper No. 28, Legal Infrastructure Related to Uranium Mining in the San Juan Basin, United States Department of Interior.

Table 1.10 Requirements to obtain rights to mine ore by federal, state, and private lands

Land Category	Requirements
Federal:	
Tribal	Secretary of Interior has general authority for leases, 25 USC 396a. Tribe must approve. Leases given by bid. Approval of Secretary of Interior required, 25 CFR 171.2. Tribe may negotiate lease if Secretary grants permission. Secretary has discretion to reject lease negotiated by Tribe, 25 CFR 171.2. Secretary may issue charter of incorporation to Tribe which may include authority for Tribe to negotiate mining leases without approval.
Allotted Indian	Leases given by bid. If Secretary of Interior approves, leases may be negotiated by Indian owners, but negotiated lease subject to rejection by Secretary, 25 CFR 172.4 and 172.6. Approval of allottee required. If patented, treat same as private land.
Public Domain	No lease required. Location of mineral deposit (staking a claim) after mineral has been discovered, 43 CFR 3831.1 and 3841.3. File locations with BLM and in accordance with 43 USC 1744. Also record in accordance with State law. Obtain patent for land claimed, 30 USC 29, 43 CFR Part 3860. Mill sites may be claimed by location and patenting, 30 USC 42, 43 CFR Subpart 3844. If claim has been patented, treat same as private land.
Acquired Public	Mineral estate on acquired lands can be leased by BLM, 43 CFR 3501.3-1, subject to exceptions (43 CFR 3501.1-5 and 3501.2-1). Permittee who prospected and discovered is entitled to preference right lease, 43 CFR 3520.1-1(a)(3). BLM leases land which contains valuable minerals on competitive basis, 43 CFR 3520.1-2(a). If land is not under BLM jurisdiction, consent of governmental entity having jurisdiction is required before lease issues.
Withdrawn Public	For public domain land withdrawn for power development, laws are same as for land in Public Domain, 30 USC 621. If withdrawal does not preclude mining, BLM can lease mineral estate. Agency having jurisdiction of withdrawn land reports any objections to mining activity, based on land use for which withdrawal was made. If controlling agency recommends stipulations in lease, they are included, 43 CFR 3501.3-1(a)(c). Leases from Dept. of Energy on lands withdrawn for DOE use under 42 USC 2097.

Table 1.10 (Continued)

Land Category		Requirements
6.	Reserved Public	Some Federal lands are disposed of with minerals reserved to the government; see e.g. 43 USC 299, 43 CFR 3814.1 and 30 USC 50. For these lands, lease issued by BLM requires conformance with law under which reservation was made, 43 CFR 3501. 3-2(2). For lands reserved or segregated for particular purpose, special requirements may be made by BLM for protection and use of land for purpose that it was reserved or segregated.
7.	National Forest System	Public Domain land inside National Forest System boundaries are subject to General Mining Law of 1872, with the following exceptions: 36 CFR 252.1, (a) If Department of Agriculture requires operations plan, it must be submitted. Department of Agriculture approves plan; (b) Operations must minimize environmental impact on surface resources on System Lands, 36 CFR 252.8; (c) Surface reclamation required, 36 CFR 252.8(g); (d) Inspection and compliance with plan responsibility of Department of Agriculture, 36 CFR 252.7. Acquired National Forest System Land same as Acquired Public Land.
8.	State	Leases obtained from appropriate State Agency according to state law.
9.	Private	Lease of mineral estate (or total estate) by private negotiation.

Source: San Juan Basin Regional Uranium Study, Working Paper No. 28, Legal Infrastructure Related to Uranium Mining in the San Juan Basin, United States Department of Interior.

Table 1.11 Requirements for mining and environmental plans by federal, state, and private lands

Land Category	Requirements
Federal:	
Tribal	Mining plan must be approved by USGS. If lease requires revegetation, the revegetation work is included in mining plan. Mining plan can be changed by mutual consent of USGS and operator, 25 CFR 177.6. BIA evaluates environmental effect of proposed operations and formulates environmental mitigation requirements. BIA consults with USGS, 25 CFR 177.4.
Allotted Indian	Same as Tribal Land, 25 CFR 177.1., unless allotted land has been patented. If patented, treat same as private land.
Public Domain	Plan same as acquired public land.
Acquired Public	Geological survey approval of mining plan to mitigate adverse environmental effects for federal leases, 30 CFR 231.10.
Withdrawn Public	Stipulations can be put in the lease by the agency for whom the land was withdrawn. These could affect operations but no formal submission of plans required, 30 CFR 231.10.
Reserved Public	Lessee must conduct operations in conformance with such requirements as may be made by BLM. Requirements will conform to purposes for which land was reserved, 43 CFR 350.3-2(b). Approval of mining plan required, 30 CFR 231.10.
National Forest System	Operations plan submitted to District Ranger, Department of Agricultural, if he deems it necessary, 36 CFR 252.4. Reclamation of surface required under opertor's plan, 36 CFR 252.8(g). Compliance with Federal and State environmental laws, preserve scenic values, wildlife, etc., 36 CFR 252.8. District Ranger, Department of Agriculture, inspects and assures compliance with operations plan, 36 CFR 252.7.
State	Mine plan filed with and approved by State.
Private	Same as State Land.

Note.--Some states require submission of mining and reclamation plans for all land.

Source: San Juan Basin Regional Uranium Study, Working Paper No. 28, Legal Infrastructure Related to Uranium Mining in the San Juan Basin, United States Department of Interior.

1.4.2. State Regulations

Federal statutes and regulations control many areas of environmental quality. Most state licensing or regulatory authority is often the result of a Federal-State agreement. However, land reclamation for uranium mining on federal and nonfederal lands is principally under state control. Table 1.12 shows the regulatory scheme for six states with uranium mining, and Appendix D lists the specific laws, regulations, and guides indicated generally in Table 1.12.

Agreement states have made formal arrangements with the NRC to develop programs to issue by-product, source material, and processing licenses. The Atomic Energy Act (Sec. 274), as amended, requires agreement states to provide by 1981 regulatory programs that are equivalent to or more stringent than the federal requirements for mill operations. Much of the environmental regulation of mining operations outside of federally controlled lands, especially for reclamation activities, currently depends upon state or local requirements. No NRC licenses are required for mining, except in situ.

The Federal Water Pollution Control Act amendments of 1972 give EPA National Pollution Discharge Elimination System (NPDES) permitting authority. However, Section 402 provides for approval of a state or interstate program to permit. The Administrator has established guidelines specifying procedural and other elements that must be present to obtain approval (40 CFR 124). Where states have not been approved, applicants apply for discharge permits from EPA. However, EPA asks what state requirements should also be certified so that state standards are met. Column 2 of Table 1.12 lists states that are approved to issue NPDES permits.

The Clean Air Act (CAA) amendments of 1970 and 1977 require, under Section 110, that State Implementation Plans (SIP's) must be submitted for approval to EPA for implementation of CAA on a local level. The approval and implementation of State plans are given in 40 CFR 52. In areas where NAAQS are violated, SIP's must produce compliance by 1982. If a state fails to enforce its plan, EPA may enforce it. There are currently no emission standard regulations specific for uranium mining by State governments.

Table 1.12 State laws, regulations, and guides for uranium mining

	General			Mining							
				Permits		Environmental Quality					Health and Safety
							Water		Land		
State	NRC Agreement State	NPDES Permit State	Water Use	Exploration Rights	Mining Rights	Air	Surf	UG	Solids	Reclam	
COLORADO	Yes	Yes	-	-	-	-	-	-	-	-	-
Department of Health											
Water Quality Control Div.	-	-	-	-	-		x	x	-	-	-
Air Quality Control Div.	-	-	-	-	-	x	-	-	-	-	-
Department of Natural Resources											
Div. of Water Reserves (State	-	-	x	-	-	-	-	-	-	-	-
Board of Land Commissioners	-	-	-	x	x	-	-	-	-	x	-
Mined Land Reclam Bd	-	-	-	x	x	-	x	x	-	x	-
Division of Mines	-	-	-	-	-	-	-	-	-	-	x
NEW MEXICO	Yes	No	-	-	-	-	-	-	-	-	-
State Land Commission	-	-	-	x	x	-	-	-	-	-	-
Dept. of Energy and Minerals	-	-	-	-	-	-	-	-	-	x	x
Dept. of Natural Resources	-	-	x	-	-	-	-	-	-	-	-
Env. Improvement Div.	-	-	-	-	-	x	x	x	x	-	x
TEXAS	Yes	No	-	-	-	-	-	-	-	-	-
Dept. of Water Resources	-	-	x	-	x	-	x	-	x	x	-
R.R. Commission of Texas	-	-	-	x	x	x	x	x	-	x	-
General Land Office	-	-	-	x	x	-	-	-	-	x	-
Dept. of Health	-	-	-	-	-	-	x	-	-	-	x
Air Control Board	-	-	-	-	-	x	-	-	-	-	-
UTAH	No	No	-	-	-	-	-	-	-	-	-
State Engineer	-	-	x	-	-	-	-	-	-	-	-
Dept. of Social Services	-	-	-	-	-	-	-	-	-	-	-
Division of Health	-	-	-	-	-	x	x	x	-	-	x
Water Pollution Control Bd.	-	-	-	-	-	-	x	x	-	-	-
Dept. of Natural Resources	-	-	-	x	x	-	-	-	-	x	-

Table 1.12 (continued)

	GENERAL			Mining							
				Permits			Environmental Quality				Health and Safety
							Water		Land		
State	NRC Agreement State	NPDES Permit State	Water Use	Exploration Rights	Mining Rights	Air	Surf	UG	Solids	Reclam	
WASHINGTON	Yes	Yes	-	-	-	-	-	-	-	-	-
Dept. of Natural Resources	-	-	-	x	x	-	-	-	x	x	-
Dept. of Ecology	-	-	x	-	-	-	-	-	-	-	-
Office of Water Programs	-	-	-	-	-	-	x	(No)	-	-	-
Dept of Social Services & Health	-	-	-	-	-	-	-	-	-	-	-
Health Services Division	-	-	-	-	-	-	-	-	-	-	x
Air Quality Division	-	-	-	-	-	x	-	-	-	-	-
WYOMING	No	Yes	-	-	-	-	-	-	-	-	-
State Inspector of Mines	-	-	-	x	x	-	-	-	-	x	x
State Engineers Office	-	-	x	-	-	-	-	-	-	-	-
Dept. of Env. Quality	-	-	-	-	-	-	-	-	-	-	-
Air Quality Div.	-	-	-	-	-	x	-	-	-	-	-
Water Quality Div.	-	-	-	-	-	-	x	x	-	-	-
Land Quality Div.	-	-	-	x	x	-	-	-	x	x	-
Solid Waste Management	-	-	-	-	-	-	-	-	x	-	-

Note.--An "x" indicates the existence of one or more controlling laws, regulations, or guides. See Appendix D for a list of the specific laws, regulations, or guides.

Applicable laws, regulations, and guidelines that apply to uranium mining in Colorado, New Mexico, Texas, Utah, Washington, and Wyoming are discussed below. Laws and regulations of other previously mined or potential uranium mining states, such as Arizona, California, Idaho, Montana, and South Dakota, are not reviewed. However, the basic environmental considerations of uranium mining should not be significantly different for other states.

1.4.2.1. Colorado

Colorado is an NRC "Agreement State" and has been approved by the EPA to issue NPDES discharge permits. Both radiation and water quality regulatory activities are under the jurisdiction of the Colorado Department of Health. The Health Department's Radiation and Hazardous Wastes Control Division administers radiation control activities and the control of hazardous wastes disposal. However, there are no operable rules or regulations for mining. Water quality is the responsibility of the Water Quality Control Commission (affiliated with the Health Department), which promulgates water quality standards and control regulations, and the Health Department's Water Quality Control Division, which administers and enforces the Commission's regulations and issues NPDES permits, as well as being responsible for numerous other water quality activities.

Colorado's permitting of discharges to "navigable" waters has been approved by EPA. Unlike most states, Colorado has promulgated specific "Guidelines for Control of Water Pollution from Mine Drainage" (November 10, 1970). These guidelines have the status of regulation since the State does not issue the NPDES permit unless the guidelines will be met. Colorado also has "Rules for Subsurface Disposal Systems" that, in conjunction with other rules, may assure protection of groundwater. These "Rules" cover all wastes that are disposed of underground, whether by direct or indirect means. "Wastes" include "any substance, solid, liquid, or gaseous, including radioactive particles thereof, which pollute or may tend to pollute any waters of the State." Solid waste and other land disposals are covered by Section 25-8-501, CRS 1973, as amended. In cases where these regulations do not control, the rules for subsurface disposal systems may apply.

The Colorado Department of Natural Resources administers water use in Colorado. As with most Western States, water is not in great abundance in Colorado. Determination of priority of water rights to surface and tributary groundwater is under the jurisdiction of a system of Water Courts, while the Division of Water Resources (State Engineer) administers and controls the allocation of actually available waters on an annual basis according to water rights priorities.

There are no State air quality standards or regulations that apply specifically to uranium mining. However, the National Ambient Air Quality Standards and various State emission control regulations apply to uranium mining activities as they do to all other types of emission sources. Colorado's air quality activities are the responsibility of the State Health Department's affiliated Air Quality Control Commission and its Air Quality Control Division. The Commission defines State air quality policy and promulgates air quality ambient standards and emission control regulations, while the Division administers and enforces the air quality regulations and issues emission permits.

The Board of Land Commissioners, affiliated with the Colorado Department of Natural Resources, issues permits for prospecting and controls leases for mining on State lands. The Board has policies and regulations concerning environmental impacts on prospected or leased lands.

The Colorado Mined Land Reclamation Board was created in 1976. It is adminstered by the Department of Natural Resources. The Board issues permits for all mining operations on all Federal and non-Federal lands in the State. The stated intent for Colorado Mined Land Reclamation Law is "to allow for the continued development of the mining industry in this State, while requiring those persons involved in mining operations to reclaim land affected by such operations so that the affected land may be put to a use beneficial to the people of this State. It is the further intent...to conserve natural resources, aid in the protection of wildlife and aquatic resources, and establish agricultural, recreational, residential, and industrial sites and to protect and promote the health, safety, and general welfare of the people...." The Board has established rules and regulations to implement the law. Rule 5 (Prospecting Notice and Reclamation Requirements) considers prospecting a separate activity, but still covered by certain reclamation requirements. The reclamation performance standards of Rule 6 have specific requirements for grading, hydrology and water quality, wildlife safety and protection, topsoiling, and revegetation. Rule 7 ("Surety") assures reclamation. Before the Board issues any permit and before any Notice of Intent to Prospect is valid, the applicant must post surety with the Board. The amount of surety, established by the Board, is to be sufficient to fully reimburse the State for all expenses it would incur in completing the reclamation plan in the event of default by the operator.

Colorado also has regulations that apply for health and safety in mining operations. For each invidual employee of any mining operation within the state a lifetime history is maintained on exposure to radon daughter concentrations when certain minimum values are reached. The State Department of Natural Resources Division of Mines administers these.

1.4.2.2 New Mexico

In New Mexico, a mine plan must be filed with and approved by the State Mining Inspector before he will issue a permit. The State Mining Inspector does not review the plan for environmental impact. Groundwater use rights are established by the State Engineer, and the Land Commission handles exploration and mining rights. The engineer's office issues a permit for beneficial use of any water pumped from uranium mines. However, the Navajo tribe claims jurisdiction of the State's groundwater in the northwest region of New Mexico. It is likely that the Departments of Interior and Justice will eventually become involved in this dispute as Trustees for the tribe.

Approval status has been given, with some exceptions, by EPA to New Mexico's plan for the attainment and maintenance of national air standards (40 CFR 52.1622). However, neither Federal nor State regulations include specific emission standards for uranium mining. But "Ambient Air Quality Standards" (40 CFR 50.6) on suspended particulates apply to all sources of air pollution.

New Mexico is an NRC agreement state, but it is not an approved NPDES state. Part 2 of the amended Water Quality Control Commission regulations applies to any discharge that is not subject to a permit under the NPDES system. The State requires approved discharge plans for discharges that could contaminate groundwater. However, the applicable NPDES regulations (Subpart E-Uranium, Radium and Vanadium Ores Subcategory, 40 CFR 440.50) have been challenged by some mine operators. They claim that discharges to a dry arroyo do not constitute "the discharge of pollutants into the navigable water, water of the continguous zone, and the oceans." Because more than half of active New Mexico mine discharges have NPDES permits that are now under adjudication, there is no enforcement and discharges may not be in accordance with Standards. If the NPDES challenge is sustained, then New Mexico's Part 2 regulations could be applied, even though they are not particularly suitable for uranium mining discharges. Possibly only the regulations on chemical oxygen demand and settling of heavy metal solids would apply to uranium mine wastes. The Part 3 "Regulations for Discharges onto or below the Surface of the Ground" (3-100) that are designed to "protect all groundwater" would also be important. A discharge plan is required for effluent discharges that move directly or indirectly into groundwater, if the effluent contains any of the contaminants listed in Section 3-103 a, b, and c, or toxic pollutants. Since the list of contaminants includes uranium and radium, New Mexico can approve only discharge plans meeting the drinking water standards.

There are no state regulations for solid wastes and land reclamation for mining operations. The mining plan and bonding requirements associated with mining permits determine the extent of mining reclamation.

Radiation safety requirements (Sections 74-3-1 et seq NMSA 1978) apply to both mining and milling. Air quality monitoring in underground mines currently involves potential duplication of effort by the New Mexico Mine Inspector (69-5-7 et seq NMSA 1978) and the Federal Mine Safety and Health Administration (30 CFR 57.5-37).

1.4.2.3 Texas

Texas is an NRC agreement state, but not an EPA approved NPDES permit state. The Department of Water Resources controls water use. Even though much of the water used in Texas comes from wells, there are no regulations on pumping groundwater. However, some counties have regulations that limit groundwater withdrawal to control subsidence.

Specific regulations for in situ uranium mining are enforced by the Texas Department of Health (TDH). Since Texas is an agreement state, its regulations reflect all appropriate NRC regulations. The TDH also implements the Safe Drinking Water Act (SDWA) and monitors groundwater to assure that its provisions for radium and selenium concentrations are met.

The General Land Office (GLO) issues prospecting permits and mining leases on state-owned lands. Mining and reclamation plans for uranium mining on state-owned lands are reviewed for approval by GLO. The "Texas Uranium Surface Mining and Reclamation Act" exempts state-owned lands from regulation by the Railroad Commission.

Surface mining is regulated by the Railroad Commission. All requirements of state and federal laws must be fulfilled before a permit is issued. Mining and reclamation plans must be submitted and approved. A bond is required to assure reclamation after mining.

The Texas Air Control Board administers provisions of the Clean Air Act. Except for suspended particulates, there are no applicable standards, i.e., there are no state source standards, for uranium mining.

The Texas Guides and Regulations for Control of Radiation (TRCR) do not apply to surface uranium mining. They do apply to in situ mining due to NRC agreement state licensing. The radioactive content of water discharged from all mines to the environment must not exceed TRCR limits.

1.4.2.4 Utah

Utah is neither an NRC agreement state nor an NPDES permit approved state. The Utah State Engineer's Office is responsible for approval of water use rights. The Department of Natural Resources oversees exploration and mining rights on State lands.

The Division of Oil, Gas, and Mining of the Department of Natural Resources issues permits for uranium mining operations, except in situ mining licensed by the NRC. A mining and reclamation plan must be approved. Rule M-10 standards include consideration of land use, public safety and welfare, impoundment, slopes, high walls, toxic materials, roads and pads, draining, structures and equipment, shafts and portals, sediment control, revegetation, dams, and soils. Bonding requirements assure reclamation.

Discharges to surface waters are regulated under the EPA administered NPDES system and the Utah Water Pollution Committee. Utah does have separate regulations administered by the Department of Social Services. These are applied to mining operations such as non-discharging waste water systems and in situ mining where no NPDES permit is required.

No sources of pollution will be allowed to cause groundwaters to exceed drinking water standards. The applicable standards for classes 1A and 1B domestic water sources are given in Wastewater Disposal Regulations, Part II.

Utah is developing radiation safety regulations. We do not expect that they will apply to uranium mining, since they are based on the model state suggested regulations.

1.4.2.5 Washington

Washington is an NRC agreement state and an NPDES approved permit state. The Department of Ecology regulates water use and water quality. Washington has no regulations for groundwater. These waters could be protected under the Safe Drinking Water Act.

The Department of Natural Resources controls exploration and mining rights for state-owned lands only. The mineral lease law covers both surface and underground mining but not in situ or heap leaching. The State Reclamation Act applies to state and private lands only. A mining and reclamation bond is required before a permit is issued. Reclamation is assured through bonding requirements.

Washington has a Clean Air Act under which regulations have been promulgated consistent with the Federal Clean Air Act. No source emission standards have been issued for uranium mining. National Ambient Air Quality Standards could apply to suspended particulates.

Washington has rules and regulations for radiation protection, but they do not apply to uranium mining.

1.4.2.6 Wyoming

Wyoming is an approved NPDES permitting State but not an NRC Agreement State. The State Engineer's Office controls water use rights. Control is primarily on the quantity of water used, but there is some statutory responsibility regarding sedimentation. Discharges to surface waters are regulated by the Water Quality Division of the Department of Environmental Quality. The construction of any water or waste water facility requires a construction permit. Groundwater regulations have been proposed. These include groundwater quality standards for any activity. Permitting requirements specific to in situ uranium operations is one of a group of special process discharges.

The Land Quality Division of the Wyoming Department of Environmental Quality is the principle agency responsible for enforcing environmental protection standards and reclamation standards with respect to uranium mining operations. The Division also enforces mineral exploration regulations that afford protection to groundwater and restoration of significant surface disturbances.

Wyoming law requires that uranium mined land must be restored to a use at least equal to its highest previous use (W.S. 35-11-402(a)(i) and (ii)) and mining operations must be conducted to prevent pollution of waters of the State (W.S. 35-11-402(a)(vi)). Before a mining operation receives a permit it must submit to the Department a mining and reclamation plan that demonstrates compliance with the law and associated rules and regulations. The plan must contain a plan for the disposal of all acid-forming, toxic materials or materials constituting a fire, health, or safety hazard uncovered or created by the mining process: radioactive material is included (W.S. 35-11-406(b)(ix)).

An operator must also, in accord with his approved mine and reclamation plan, cover, bury, impound, contain, or dispose of toxic, acid-forming, or radioactive material determined to be hazardous to health and safety or constitute a threat of pollution to surface or subsurface waters (W.S. 35-11-415(b)(iv)). A required surety bond assures that the operator will reclaim the land according to his approved plan. If the bond is forfeited, the State is responsible for reclamation.

Wyoming has legislated authority for a position on radiological restoration of mined lands. It is described in the Division's Guideline No. 1, Section III. The Division is presently drafting regulations for radiation protection on uranium mined lands and handling of uranium mine wastes. These regulations shall set standards.

Wyoming also has a solid waste management program that presently regulates only refuse generated at mines. Solid waste disposal sites are permitted at these facilities. Solid waste regulations could be promulgated that affect mining.

In Wyoming, Ambient Air Quality Standards are applied to mining operations, and fugitive emissions are controlled to the extent that these standards are met. An Air Quality Permit is required for the construction of a uranium mining and/or processing facility, and the applicant is required to demonstrate that applicable ambient and PSD (Prevention of Significant Deterioration) provisions are met.

Wyoming has radiation protection regulations for the safety of mines while they are actually in process. These regulations are under the jurisdiction of the State Inspector of Mines. According to Wyoming Law, the protection of miners from hazardous exposure to radioactivity must conform to the American Standards Association revised Publication N 13.8, "Radiation Protection in Uranium Mines and Mills." The uranium regulations (94-R-11) are found in Chapter 3, Article 4 of Title 30 - Mines and Minerals.

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SECTION 2

INVENTORY OF URANIUM MINES

2.0 Inventory of Uranium Mines

To inventory the numbers, types, and locations of uranium mines in the United States, we used data from the Department of Energy, Grand Junction Office (DOE-GJO). We produced the inventory of uranium mines presented in this section and Appendixes E and F of this report from the DOE-GJO master data file (DOE79a) and personal communications with DOE-GJO (Ch80, ME80a, ME80b). These two sources combined yielded our own EPA master data file, which we divided into two parts - active and inactive mines.

Table 2.1 classifies active and inactive U.S. uranium properties according to the method of uranium production (mine type) based on data that were current as of 1978 (Me80a). The major mining methods are surface and underground mines (DOE79b). The remaining mining methods are only minor contributors to the total uranium ore production (DOE79b).

Table 2.1 shows a total of 340 active mines. This final total, which is 52 less than the original total of 392 active mines provided by DOE-GJO (Me80a), was derived, in consultation with DOE-GJO (Me80b), by eliminating 43 mines that were duplicated on the list and 9 that were small producers (i.e., producing only a few tons of ore for the entire year of 1978). Most (if not all) of the 52 eliminated mines were either underground or surface mines.

The original totals of 305 active underground mines and 63 active surface mines (DOE79b), whose combined total of 368 mines accounts for the later eliminated 52 mines that were duplicate listings or small producers, were the totals we used in modeling the average underground and surface mines in this study. The differences between these totals and the smaller Table 2.1 totals of 256 active underground mines and 60 active surface mines are insignificant compared with other uncertainties in predicting health effects. The smaller totals for underground and surface mines would introduce differences of less than 17% and less than 5% for the active average underground and average surface model mines, respectively.

Table 2.2 gives locations and types of active uranium mines by state. With respect to the number of mines, Colorado and Utah dominate the inventory, especially for underground mines. However, since New Mexico and

Wyoming have large mines (underground in New Mexico, and surface in Wyoming) and dominate ore production, New Mexico is the site of our model active underground mines and Wyoming is the site of our model active surface mines. Our model in situ leaching operation is also sited in Wyoming, which is one of two states mining uranium with that method. Appendix E gives a complete inventory of active uranium mines.

The numbers of inactive uranium mines according to state and mining method are given in Table 2.3. Colorado and Utah have the greatest number of inactive mines, but Arizona, Wyoming, New Mexico, and South Dakota also contain significant numbers. Since New Mexico and Wyoming have dominated ore production over the past 10 years (DOE79b), New Mexico (because of its large underground mines) is our model site for inactive underground mining and Wyoming (because of its large surface mines) is the site of our model inactive surface mine. Appendix F gives a complete inventory of inactive uranium mines.

Figures 2.1 through 2.9 are maps showing the locations, status, and types of uranium mines in Colorado, New Mexico, Texas, Utah, and Wyoming (Ch77, Co78a, Co78b, Co78c, Ea73, Gl75, Hi69, Pe79, Ut77). Since it is not always possible to show all the mines in a given district, the maps indicate only the area and number of mines in some major mining districts, particularly for Colorado and Utah. The maps do not show the location of many small mines started during the uranium boom of the 1950's because their exact locations are unknown. In Colorado alone there are over a thousand such mines.

Table 2.4 shows total ore production through January 1, 1979 for active and inactive surface and underground mines. The larger mines (>910 MT ore production) dominate the list of active mines, and the smaller mines (<910 MT ore production) dominate the inactive list. If remedial action becomes necessary for inactive mines, the information in Table 2.4 could help estimate the magnitude of such an action, at least affording a way to make rough estimates of waste rock, sub-ore, and overburden that are present at the inactive site. A recent DOE report (DOE79c) contains additional information on mining waste tonnage and acreage of specific properties.

Table 2.1 Type of U.S. uranium properties

Uranium Production Method[a]	Number of Active Properties	Number of Inactive Properties
Surface mine	60	1252
Underground mine	256	2036
Mine water production	2	1
Heap leach - dumps	1	7
Heap leach - ores	0	1
Dumps	1	42
Sub-ore	1	12
In-situ leaching	11	2
Miscellaneous	0	23
Tailings dump	2	0
Unknown	6	13
TOTAL	340	3389

(a)Categories listed in this column are modifications of the originals (DOE79a). Copper by-product and surface-underground combination categories were eliminated because they contained no properties. The miscellaneous-phosphate by-product category was reduced to miscellaneous because most phosphate by-product properties were not included in the DOE-GJO master data file (DOE79a). The low grade or protore category was changed to sub-ore to be consistent with the rest of this report.

Table 2.2 The location and type of active uranium properties

State	Surface Mine	Underground Mine	Mine Water Production	Heap-Leach Dumps	Heap-Leach Ores	Dumps	Sub-ore	In-Situ Leaching	Miscellaneous	Tailings Dump	Unknown	Total
Arizona	1	1	0	0	0	0	0	0	0	0	0	2
Colorado	5	106	0	0	0	0	0	0	0	1	3	115
New Mexico	4	35	2	0	0	1	0	0	0	0	0	42
Texas	16	0	0	0	0	0	0	8	0	1	0	25
Utah	13	108	0	0	0	0	0	0	0	0	3	124
Washington	2	0	0	0	0	0	0	0	0	0	0	2
Wyoming	19	6	0	1	0	0	1	3	0	0	0	30
TOTAL	60	256	2	1	0	1	1	11	0	2	6	340

Table 2.3 The location and type of inactive uranium properties

State	Surface Mine	Underground Mine	Mine Water Production	Heap-Leach Dumps	Heap-Leach Ores	Dumps	Sub-ore	In-Situ Leaching	Miscellaneous	Tailings Dump	Unknown	Total
Alaska	0	1	0	0	0	0	0	0	0	0	0	1
Arizona	135	189	0	1	0	0	0	0	0	0	1	326
California	13	10	0	0	0	0	0	0	0	0	0	23
Colorado	263	902	0	0	0	35	1	0	10	0	6	1217
Florida	0	0	0	0	0	0	0	0	1	0	0	1
Idaho	2	4	0	0	0	0	0	0	0	0	0	6
Minnesota	0	0	0	0	0	0	0	0	1	0	0	1
Montana	9	9	0	0	0	0	0	0	0	0	0	18
Nevada	9	12	0	0	0	0	0	0	0	0	0	21
New Jersey	0	1	0	0	0	0	0	0	0	0	0	1
New Mexico	34	142	1	0	0	0	8	0	1	0	2	188
N. Dakota	13	0	0	0	0	0	0	0	0	0	0	13
Oklahoma	3	0	0	0	0	0	0	0	0	0	0	3
Oregon	2	1	0	0	0	0	0	0	0	0	0	3
S. Dakota	111	30	0	0	0	0	0	0	0	0	0	141
Texas	38	0	0	0	1	0	0	2	0	0	1	42
Utah	378	698	0	0	0	7	1	0	6	0	3	1093
Washington	13	0	0	0	0	0	0	0	0	0	0	13
Wyoming	223	32	0	6	0	0	2	0	2	0	0	265
Unknown	6	5	0	0	0	0	0	0	2	0	0	13
TOTAL	1252	2036	1	7	1	42	12	2	23	0	13	3389

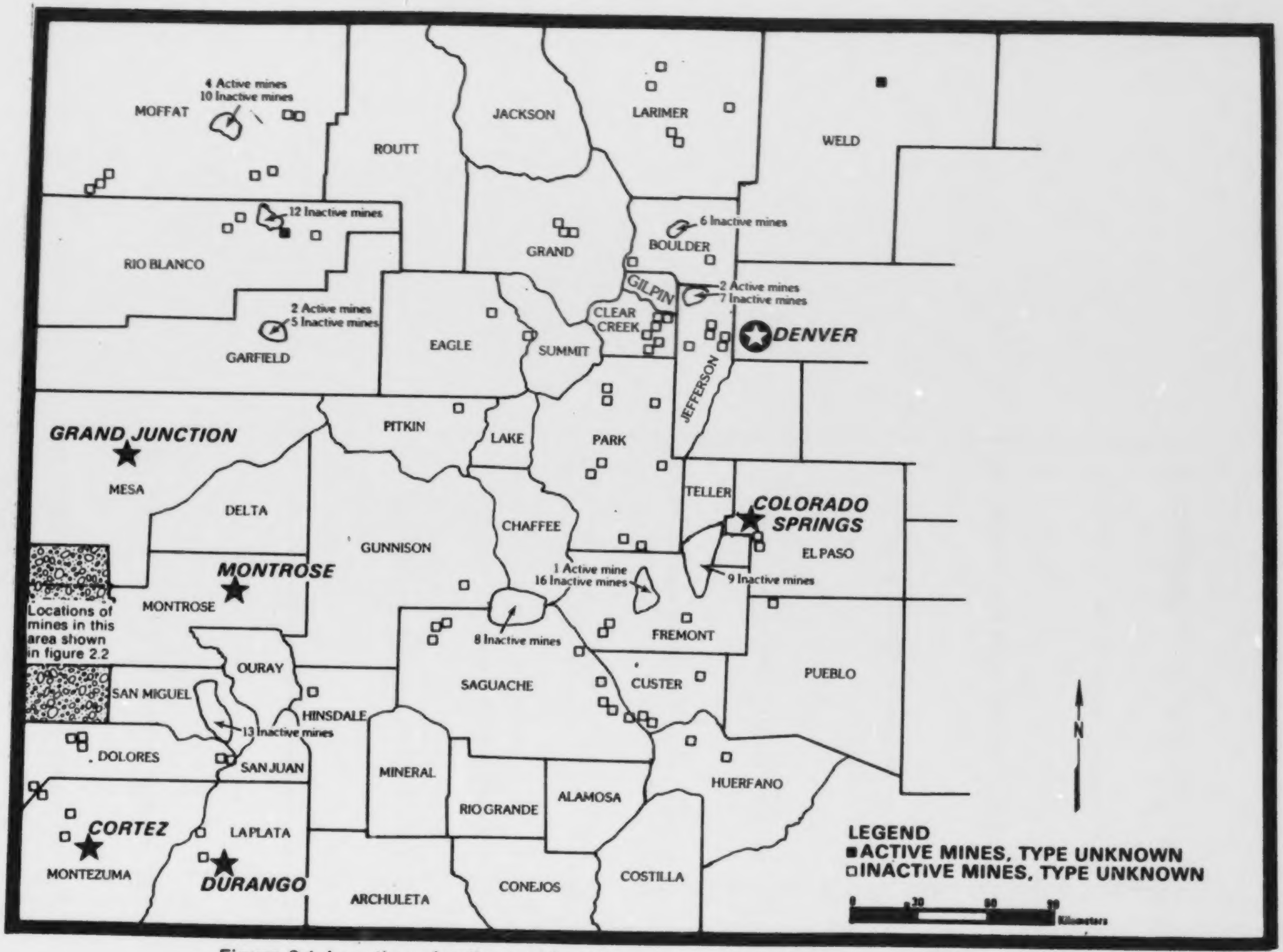

Figure 2.1 Location of active and inactive uranium mines and principal uranium mining districts in Colorado

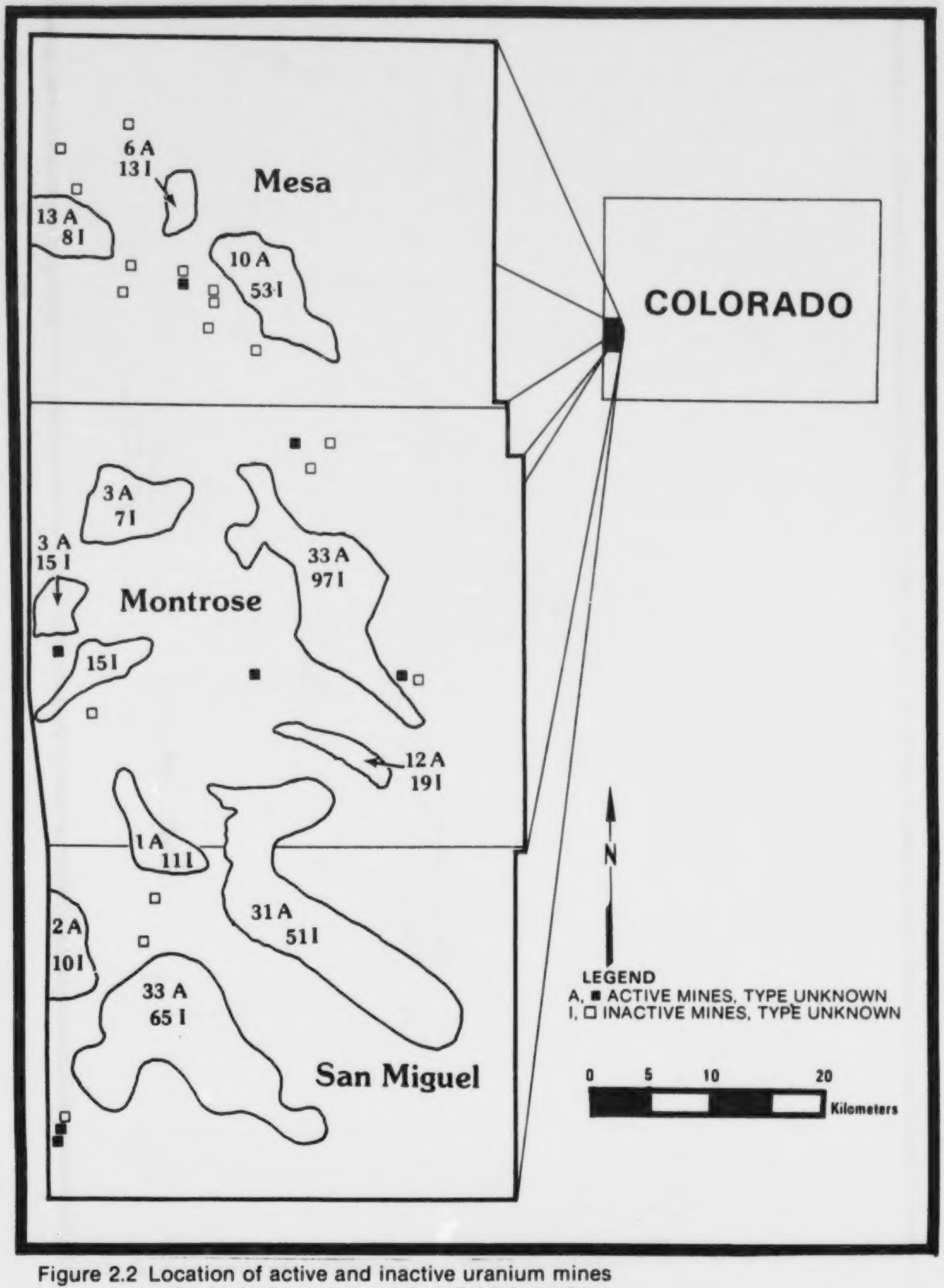

Figure 2.2 Location of active and inactive uranium mines and principal uranium mining districts in the Uravan Mineral Belt of western Colorado

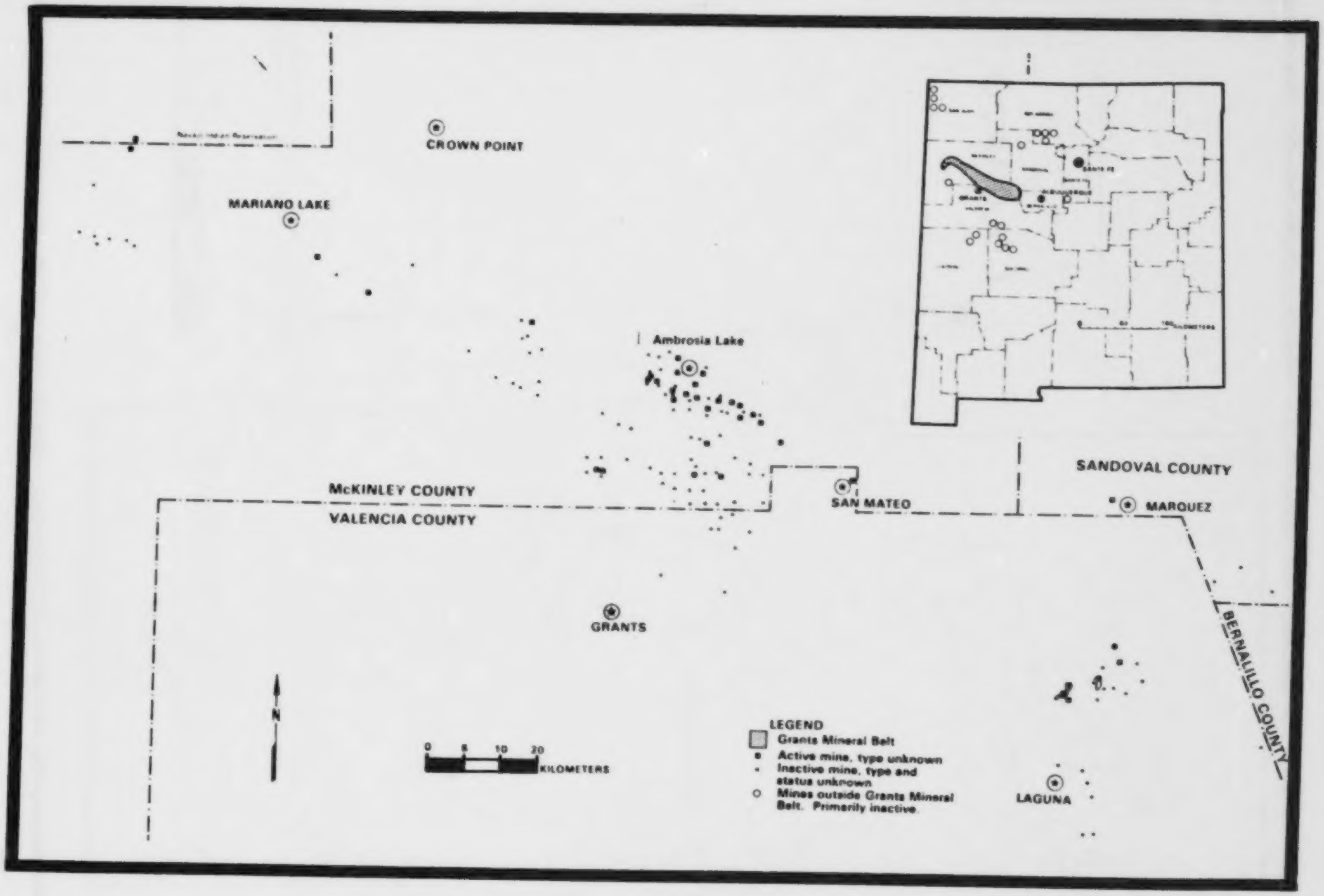

Figure 2.3 Location of active and inactive uranium mines in the Grants Mineral Belt and other areas of New Mexico

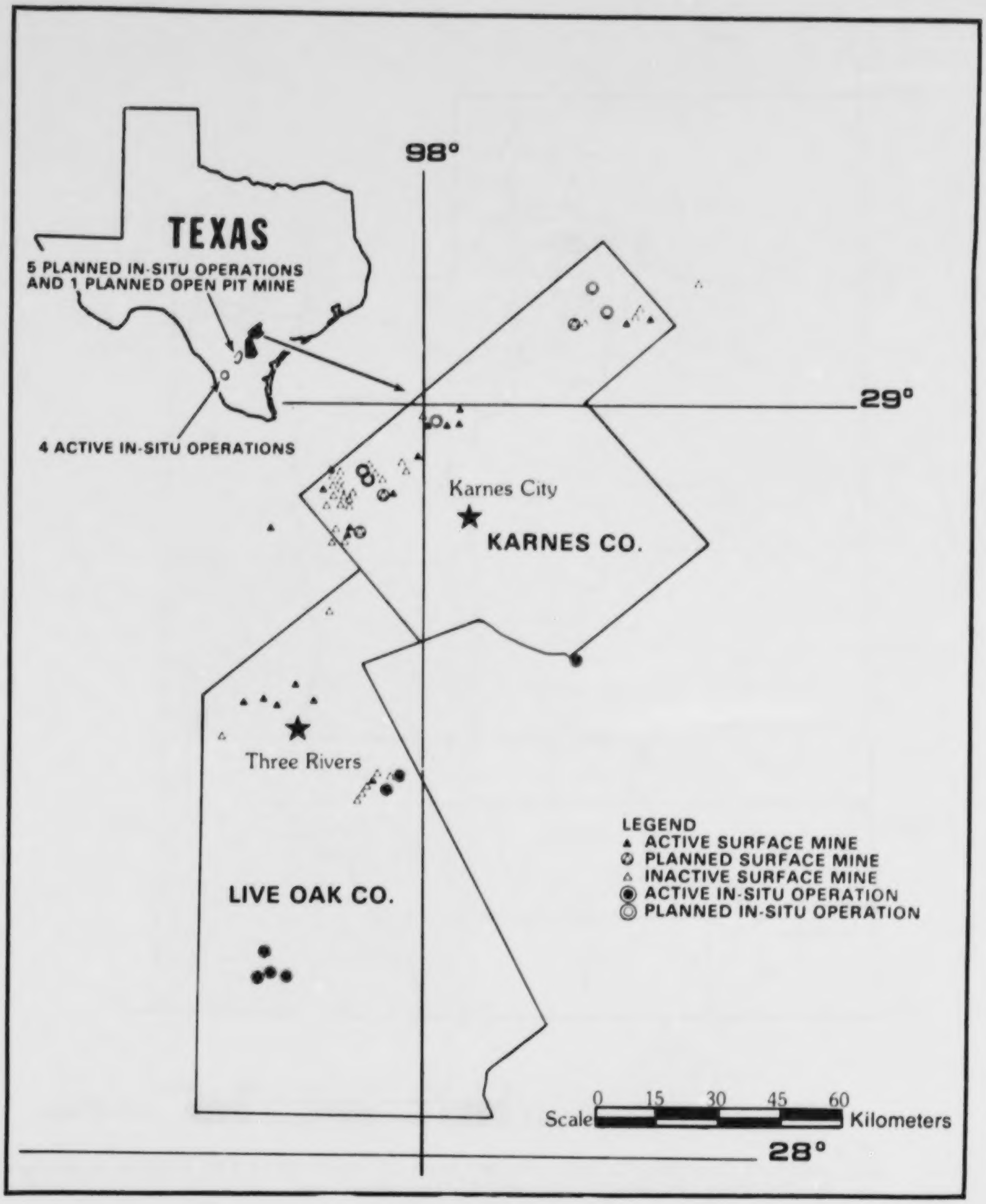

Figure 2.4 Location of active, inactive, and proposed surface and in situ uranium mines in Texas

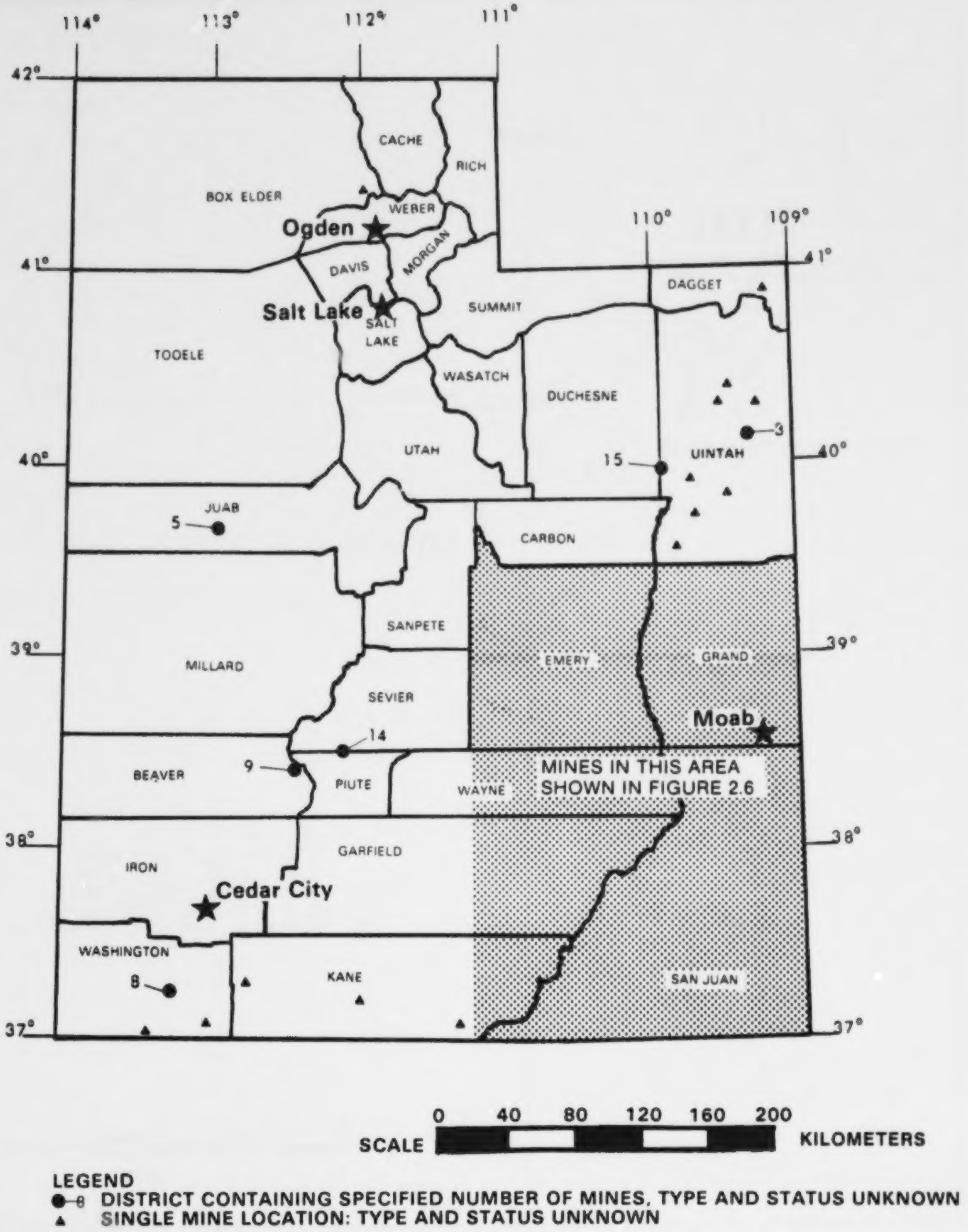

Figure 2.5 Location of uranium mines and mining districts in Utah

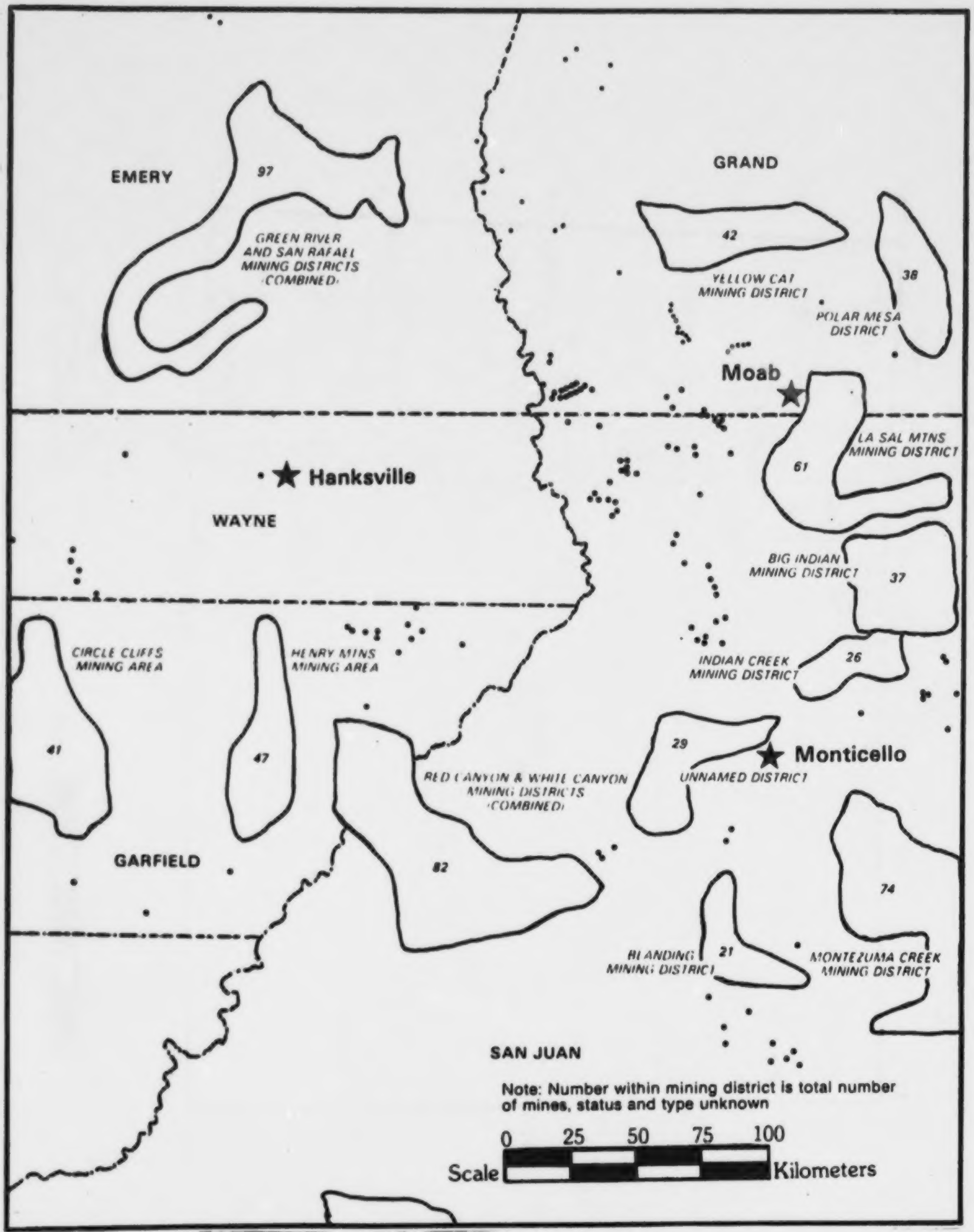
EMERY
97
GREEN RIVER AND SAN RAFAEL MINING DISTRICTS (COMBINED)
GRAND
42
YELLOW CAT MINING DISTRICT
38
POLAR MESA DISTRICT
Moab
LA SAL MTNS MINING DISTRICT
61
Hanksville
WAYNE
BIG INDIAN MINING DISTRICT
37
CIRCLE CLIFFS MINING AREA
HENRY MTNS MINING AREA
INDIAN CREEK MINING DISTRICT
26
41
47
29
Monticello
RED CANYON & WHITE CANYON MINING DISTRICTS (COMBINED)
UNNAMED DISTRICT
GARFIELD
82
74
BLANDING MINING DISTRICT
21
MONTEZUMA CREEK MINING DISTRICT
SAN JUAN
Note: Number within mining district is total number of mines, status and type unknown
0
25
50
75
100
Scale
Kilometers

Figure 2.6 Location of uranium mines and principal uranium mining districts in southeastern Utah

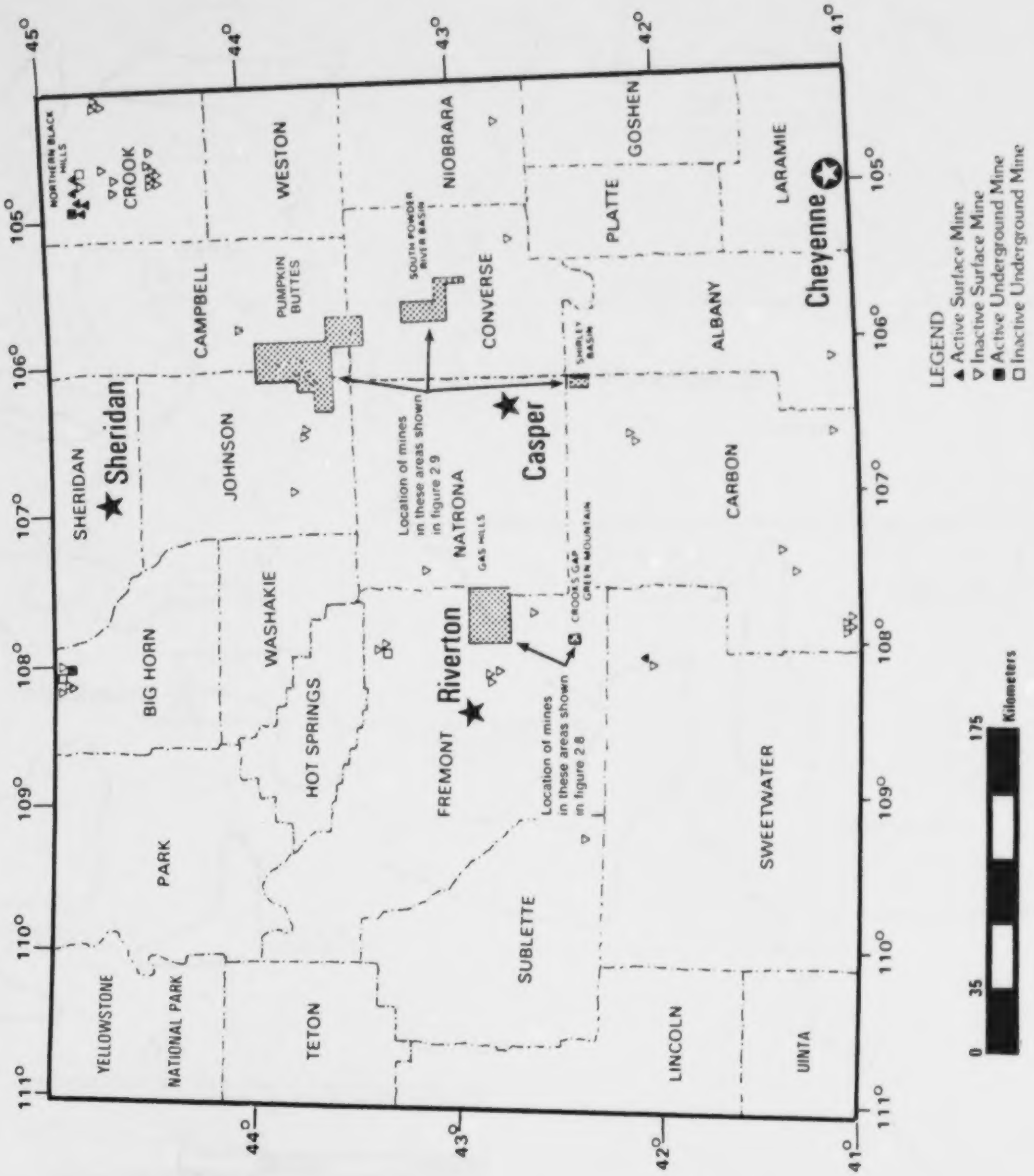

Figure 2.7 Location of active and inactive uranium mines and principal uranium mining areas in Wyoming

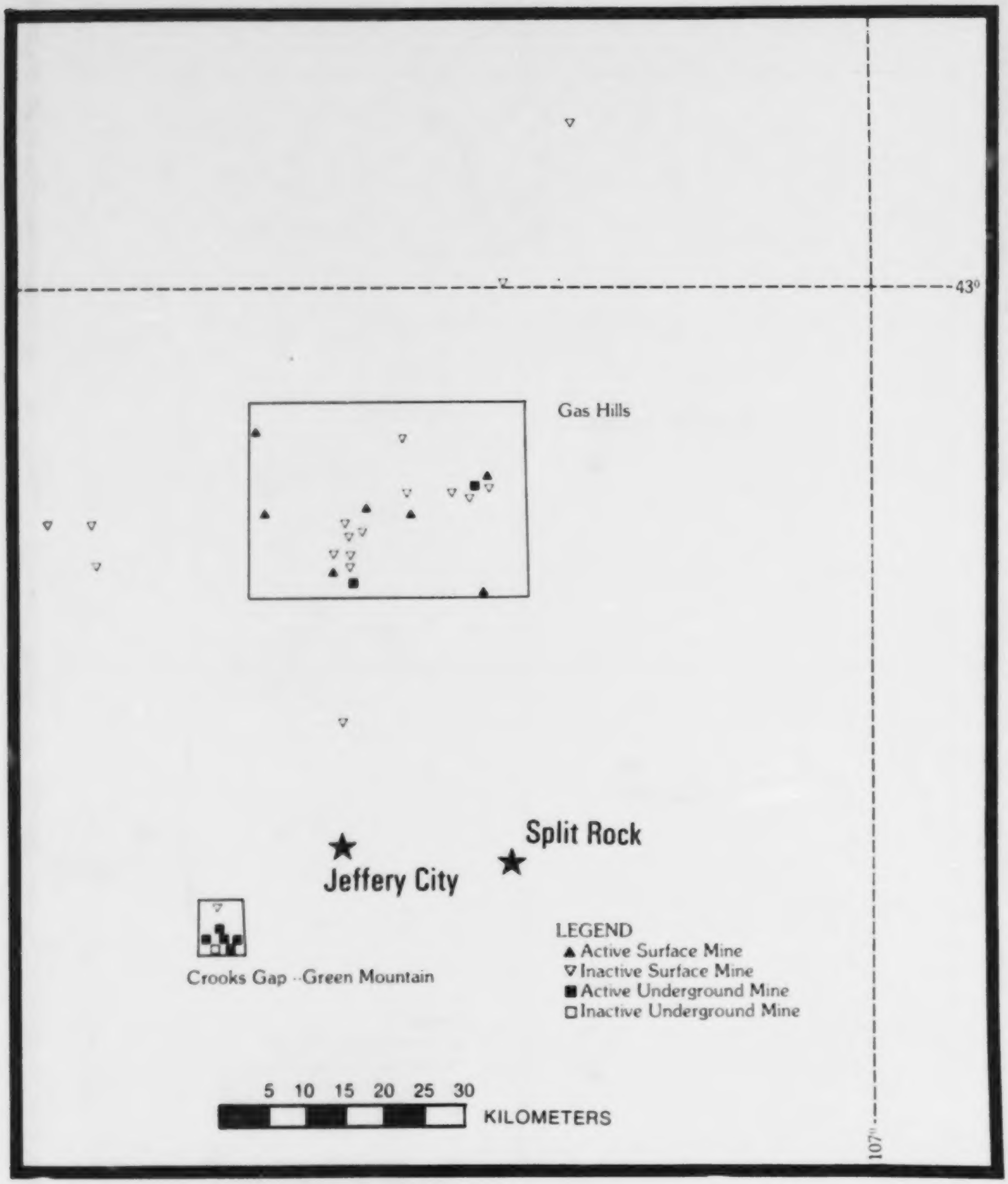

Figure 2.8 Location of active and inactive uranium mines in the Gas Hills and Crooks Gap-Green Mountain areas of central Wyoming

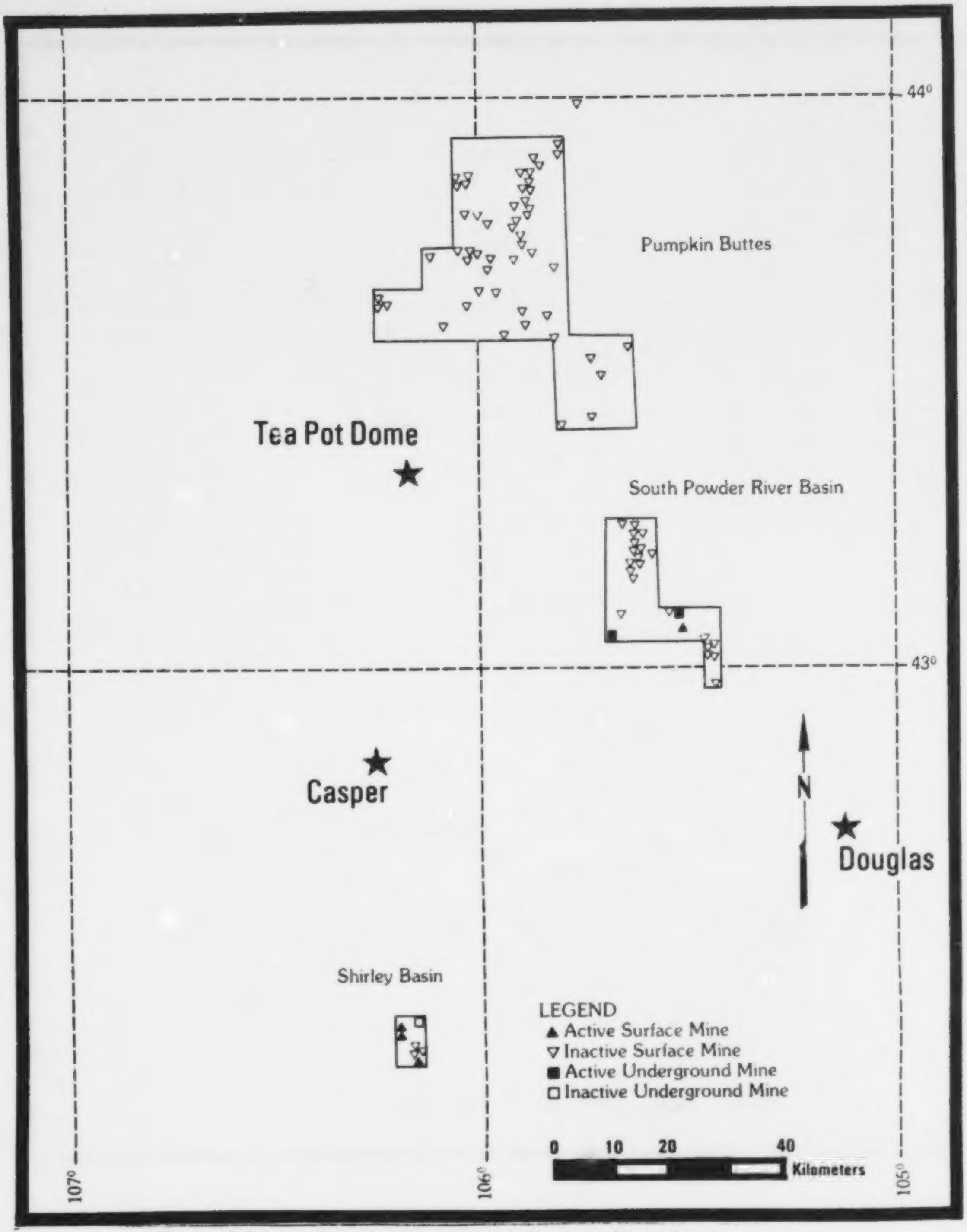

Figure 2.9 Location of active and inactive uranium mines in the Shirley Basin, South Powder River Basin, and Pumpkin Buttes areas of Wyoming

Table 2.4 Cumulative Ore Production
through January 1, 1979

Ore Production MT	Active				Inactive			
	No. Mines	(% of total)	Surface	Under-ground	No. Mines	(% of total)	Surface	Under-ground
< 91	16	(4.7)	8	3	1553	(45.8)	899	628
91-910	33	(9.7)	5	24	753	(22.2)	134	588
910-91,000	188	(55.3)	15	165	986	(29.1)	180	766
> 91,000	103	(30.3)	32	64	97	(2.9)	39	54
Total	340	(100.0)	60	256	3389	(100.0)	1252	2036

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SECTION 3

POTENTIAL SOURCES OF CONTAMINANTS TO THE ENVIRONMENT AND MAN

3.0 Potential Sources of Contaminants to the Environment and Man

3.1 Background Concentrations of Radionuclides and Trace Metals

3.1.1 Naturally Occurring Radionuclides

Potassium-40 and radionuclides in the decay chains of uranium-238 and thorium-232 are the principal sources in the earth's crust of background radiation. Figures 3.1 and 3.2 show the uranium-238 and thorium-232 decay chains. Potassium-40 constitutes 0.0118 percent of naturally occurring potassium. Its half-life is 1.26×10^9 years and, upon decay, potassium-40 emits a 1.46 MeV gamma ray in 11 percent of its disintegrations. Table 3.1 lists the average concentration of and gamma-ray energy released by these radionuclides in one gram of rock. Table 3.2 lists the radionuclide content and dose equivalent rates from common rocks and soils. Potassium-40 and the thorium-232 decay chain each contribute about 40 percent of the dose rate at 3 feet above the ground while the uranium-238 decay chain contributes approximately 20 percent of the total dose rate.

Radon-222 occurs in the uranium-238 decay chain and has a half-life of 3.8 days. It is a noble gas and, upon decay, produces a series of short-lived, alpha-emitting daughters (see Fig.3.1). The average atmospheric radon concentration in the continental U.S. is 0.26 pCi/liter (Oa72). Under most conditions, the radon daughters contribute less than 10 percent (a few tenths of a μrem/hr) to the terrestrial external dose equivalent rate. However, inhaled radon daughters contribute a large fraction of the total dose equivalent rate to the respiratory tract: about 50 percent (90 mrem/yr) to the lung and nearly all of the dose (450 mrem/yr) to the segmental bronchioles (NCRP75).

Eighty-five percent of the surface area of the United States, and nearly all of its population, is underlain by rocks and soils of sedimentary origin. However, the correlation between the bedrock activity and the aboveground activity is not clear.

In most soils, the amount of water varies from 5 to 25 percent. The soil moisture attenuates gamma radiation from the soil. The potassium-40 dose equivalent rate can decrease by 30 percent when the soil water content increases from 0 to 30 percent (OA72). Moisture can retard the diffusion of radon into the atmosphere and reduce the exposure to airborne radon daughters. Since radon daughters account for 95 percent of the gamma-ray energy from the uranium-238 series, their accumulation in the ground increases

URANIUM — 238 DECAY SERIES

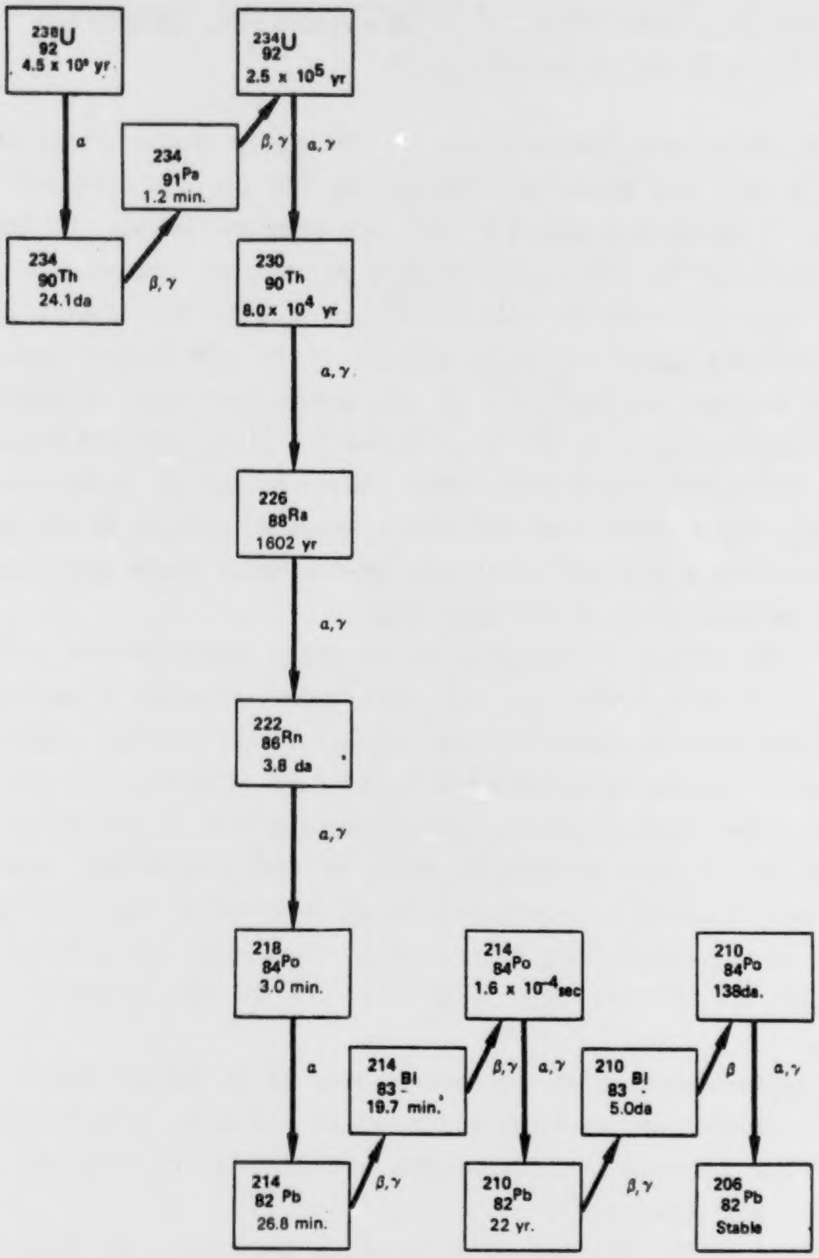

Figure 3.1 The uranium decay series showing the half lives and mode of decay.

THORIUM - 232 DECAY SERIES

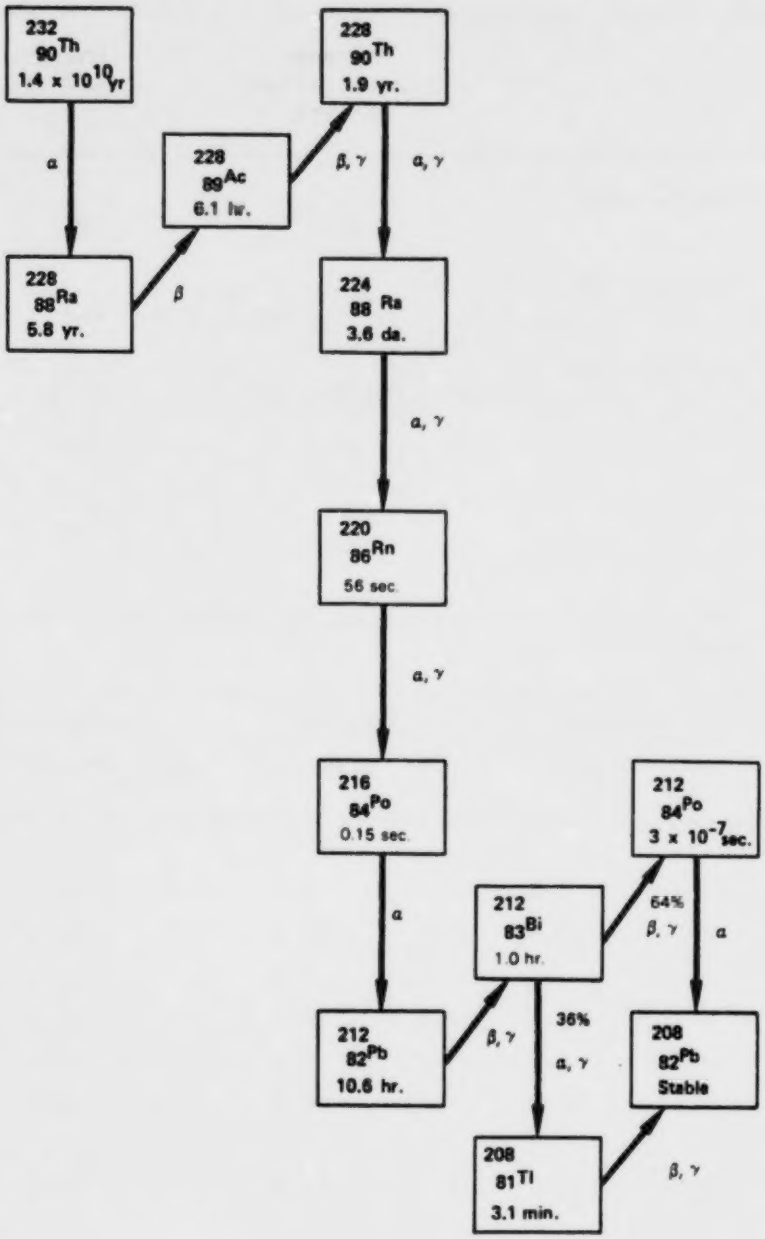

Figure 3.2 The thorium decay series showing the half lives and mode of decay.

Table 3.1 Gamma-ray energy released by one gram of rock

Isotope	Average Concentration, Percent	Energy, KeV/sec.
Uranium-238 (in equilibrium with decay products)	2.98×10^{-4}	68.2
Uranium-235 (in equilibrium with decay products)	0.02×10^{-4}	1.53
Thorium-232 (in equilibrium with decay products)	11.4×10^{-4}	87.8
Potassium-40	3.0	149
Other Elements	---	2.7

Source: Oa72.

Table 3.2 Radionuclide content and dose equivalent rates from common rocks and soil

	Uranium		Thorium		Potassium-40		Total
Rock Type	ppm	mrem/yr(a)	ppm	mrem/yr(a)	ppm	mrem/yr(a)	mrem/yr
Igneous(b) basic	0.9	5.2	2.7	7.3	1.2	14.7	27.2
Silicic (granite)	4.7	26.9	20.0	53.8	5.0	61.3	142.0
Sedimentary(b)							
Shale	3.7	21.2	12.0	32.3	3.2	39.2	92.7
Sandstone	0.45	2.6	1.7	4.6	1.1	13.5	20.7
Limestone	2.2	12.6	1.7	4.6	0.32	3.9	21.1
Upper crustal average(c)	2.8	16.0	10	26.9	2.4	29.4	72.3
U.S. surficial average(d)	1.8	10.3	9.0	24.2	1.8	21.8	56.3

(a) mrem/yr/ppm: uranium, 5.73; thorium, 2.69; potassium-40, 12.3 (Be68).

(b) Source: Cl66.

(c) Uranium and thorium averages (Ph64); potassium (He69).

(d) Source: Lo64.

the exposure from this series. Thus, soil moisture decreases the potassium-40 and thorium-232 dose equivalent rates and increases or leaves unchanged the uranium-238 series dose equivalent rate.

Snow cover also affects the terrestrial dose equivalent rate and the radon emanation rate. Gamma radiation attenuates exponentially as a function of the density and thickness of the snow cover (Oa72). However, the overall influence of snow on population exposure is negligible since, in most populated areas, there is relatively little snowfall that remains for long periods of time.

Table 3.3 shows the average dose equivalent rate due to radiation in some Western mining states (Oa72). Terrestrial radiation in the Western uranium mining states is higher than in the rest of the nation due to the greater concentration of the uranium-238 series.

Table 3.3 Average dose equivalent rates due to terrestrial radiation in western mining states

State	Terrestrial Dose, mrem/yr
Arizona	45.6
Colorado	65.8
New Mexico	51.7
South Dakota	45.6
Texas	29.0
Utah	45.6
Wyoming	45.6

Concentrations of radionuclides measured in surface and groundwater samples collected on a proposed uranium project site are listed in Table 3.4 (NRC79a). The large variations among concentrations at different collection sites are typical of surface water concentrations. (Concentrations in sea water are more uniform.) Hence, generalizations about background concentrations of radionuclides in fresh water systems are impractical. Extensive, site-specific studies over an extended period of time are necessary to obtain meaningful background concentrations for a site.

3.1.2 Stable Elements

Concentrations of metals occurring in the earth's crust generally range from several parts-per-billion (ppb) to a few parts-per-million (ppm). Measured concentrations vary widely from site to site and often in different samples taken from the same site. Table 3.5 lists the results of measurements for selected elements. It should be emphasized that these are general estimates of element composition of rocks in the United States and do not reflect large variations that occur within the different rock types.

Concentrations of metals measured in surface and groundwater samples collected from different locations on a proposed uranium project site are listed in Tables 3.6 and 3.7, respectively. There are large differences in the composition of surface and groundwaters. Table 3.8 shows the average concentrations of three trace metals that are sometimes associated with mine discharge water. These values, which were taken from the results of an extensive study (Tu69), approximate average concentrations in United States streams. Background concentrations at any specific site could be much different.

Table 3.8 Estimated average concentrations (ppb) of three metals in U.S. streams

Element	Turekian's Results	Other results from Literature
Chromium	1.4	1.0
Molybdenum	1.8	1.0
Selenium	0.2	0.2

Source: Tu69.

Table 3.4 Radionuclide concentrations in surface and groundwater in the vicinity of a proposed uranium project

Radionuclide	Concentrations, pCi/ℓ					
	Location 1	Location 2	Location 3	Location 4	Location 5	Location 6
			Surface Water			
U-238	9.8	4.0	2.5	1.8	1.6	---
Ra-226	0.4	0.5	<0.1	0.08	<0.1	---
Rn-222	145	108	42	<4	<4	---
Th-230	<0.1	0.2	0.3	<0.1	<0.1	---
Th-232	<0.1	<0.1	<0.1	<0.1	<0.1	---
			Groundwater			
U-238	2.0	4.5	2.3	3.2	3.8	1.2

Source: NRC79a.

Table 3.5 Concentrations of selected elements in igneous and sedimentary rocks, (ppm)

Element	Igneous rocks[a]	Shales[b]	Sandstones[b]	Limestone[b]
Aluminum		80,000	25,000	4,200
Arsenic	1.8	13	1	1
Boron	10	100	35	20
Barium	425	580	50	120
Cadmium	0.2	0.04	0.05	0.035
Cobalt	25	19	0.3	0.1
Chromium	100	90	35	11
Copper	55	45	5	4
Iron	56,300	47,200	9,800	3,800
Mercury	0.08	0.04	0.03	0.04
Potassium	20,900	26,600	10,700	2,700
Manganese	950	850	50	1,100
Molybdenum	1.5	2.6	0.2	0.4
Nickel	75	68	2	20
Lead	12.5	20	7	9
Antimony	0.2	1.5	0.05	0.2
Selenium	0.05	0.6	0.05	0.08
Vanadium	135	130	20	20
Zinc	70	95	16	20

(a) Source: Ta64.

(b) Source: Tu69.

Table 3.6 Concentrations of selected elements in surface water at five locations in the vicinity of a proposed uranium project

Element	Concentrations, mg/ℓ				
	Location 1	Location 2	Location 3	Location 4	Location 5
Aluminum	2.4	3.0	50	1	2
Antimony	<0.01	0.01	<0.01	<0.01	<0.01
Arsenic	<0.01	0.01	0.07	<0.01	<0.01
Barium	0.06	1.4	6.6	0.5	0.5
Beryllium	0.001	0.001	0.04	<0.001	<0.001
Cadmium	<0.001	0.001	<0.001	<0.001	0.001
Cobalt	0.01	0.01	0.10	<0.01	0.01
Chromium	0.005	0.001	0.066	0.003	0.003
Copper	0.019	0.017	0.18	0.002	0.005
Lead	<0.001	0.001	0.020	<0.001	<0.001
Mercury	0.0007	0.0008	0.0007	0.0005	0.0006
Molybdenum	0.002	0.002	0.002	0.002	0.002
Nickel	0.03	0.02	0.12	<0.01	<0.01
Selenium	0.03	0.03	<0.01	<0.01	<0.01
Vanadium	0.03	0.02	0.45	0.01	0.02
Zinc	0.04	0.05	0.35	<0.01	0.01

Source: NRC79a.

Table 3.7 Concentrations of selected elements in groundwater at six locations in the vicinity of a proposed uranium project

Element	Concentrations, mg/ℓ					
	Location 1	Location 2	Location 3	Location 4	Location 5	Location 6
Aluminum	<0.1	1	0.8	14.7	0.2	10
Antimony	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Arsenic	<0.01	<0.01	<0.01	<0.01	0.01	0.04
Barium	<0.1	0.5	0.5	0.2	0.3	0.7
Beryllium	0.001	<0.001	<0.001	0.003	0.001	<0.001
Cadmium	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Cobalt	<0.01	<0.01	<0.01	0.01	<0.01	<0.01
Chromium	<0.001	0.003	0.003	0.004	0.001	0.003
Copper	0.003	0.003	0.006	0.027	<0.001	0.006
Lead	0.003	<0.001	<0.001	0.003	<0.001	0.002
Mercury	<0.0004	0.0012	0.0005	<0.001	<0.0004	0.0017
Molybdenum	0.001	0.002	0.002	0.001	0.001	0.003
Nickel	<0.01	<0.01	<0.01	0.02	<0.01	<0.01
Selenium	0.06	<0.01	<0.01	<0.01	0.13	<0.01
Vanadium	0.01	0.01	0.02	0.01	0.02	0.08
Zinc	4.4	0.07	0.21	0.07	0.01	0.02

Source: NRC79a.

3.2 Water-Related Aspects of Uranium Mining

3.2.1 Previous and Ongoing Hydrologic and Water Quality Studies Related to Uranium Mining

In the late 1950's and early 1960's, the U.S. Public Health Service conducted field studies to determine the water quality impacts of the uranium mining and milling industry. The studies emphasized uranium milling rather than mining. The Federal Water Pollution Control Administration conducted extensive stream surveys to assess the effects of uranium milling (but not mining) on the main stem and principal tributaries of the Colorado River. Subsequent stream survey work in Colorado by the Water Pollution Control Commission and the U.S. Geological Survey (Mo74, We74) mentioned a portion of the Uravan Mineral Belt and uranium mines therein, but the work did not emphasize uranium mines. Significant amounts of acidity and total trace metal concentrations were found in streams from 18 different mining areas. Dilution and chemical precipitation below mine drainages decreased concentration and increased the pH. Given enough time and distance, the streams recover naturally, but the accumulations of trace metals in the sediments increase. Field observations in 1971-72 of streams in most of Colorado indicated that approximately 724 km of streams in 25 different mining areas were adversely affected by mine drainage (We74).

Discussions of the impacts of uranium mining on water quality or quantity are incidental in numerous impact statements and environmental reports prepared by industry and (or) the U.S Nuclear Regulatory Commission as an integral part of licensing or relicensing uranium mills. Coverage on mining is usually minor as the principal focus is on milling impacts. The same is true for the recently prepared generic EIS on regulation of uranium milling (NRC79b).

Radiochemical assessment of surface and groundwater in uranium mining districts of New Mexico is done by self-monitoring programs associated with NPDES permits. Also, radiochemical assessment studies have been funded recently by the New Mexico Environmental Improvement Division and U.S Environmental Protection Agency. Self-monitoring, particularly in the preoperational phase, characterizes mining and milling operations in all of the concerned States. These, together with results of surveys by State personnel, have resulted in extensive files of water quality data, flow measurements, field observations of mine conditions, and exchanges between industry, regulatory agencies, and the public. Rarely are the data assembled and interpreted for dissemination outside a given agency. States experiencing rapid growth in uranium mining and milling are undoubtedly placing first priority on activities directly related to licensing, monitoring, and otherwise implementing regulations. Unfortunately, there is no concerted effort to prepare broad assessments of the cumulative impacts of mining and milling. Texas, New Mexico, and Wyoming are cases in point. Critical review and synthesis of these types of data can produce rather useful information. For example, the publication "Water Quality Impacts of Uranium Mining and Milling Activities in the Grants Mineral Belt, New Mexico" (EPA75) addresses the groundwater and surface water changes as the result of extensive uranium mining and milling production in a relatively confined area.

Some states have since initiated review of their data files, conducted field studies, and, in some cases, contracted study teams to investigate similar water quality changes. For example, a recent report by the Wyoming Department of Environmental Quality summarizes 16 years of aqueous radium and uranium data. The study reports that significant amounts of Ra-226 and uranium were present in surface water in the Shirley Basin as a result of inadequate mine water treatment (Ha78).

In Texas, surface water and groundwater monitoring conducted by industry, as well as by State and Federal agencies, reveal little or no change of chemical quality attributable to uranium mining and milling (Ge77, Ka76). This conclusion is based on 586 samples collected from 198 stations over a period of 39 years but primarily from 1961 to 1975. The State monitoring program by several agencies is continuing, but either summary reports are not issued or are two years overdue, depending on the agency. Not all of the findings exonerate the industry. One survey (It75) showed that none of the mine water from the 10 lakes that were sampled was suitable for human use. The lakes were also unsuitable for irrigation due to mineralization of the water by sulfate, chloride, and TDS. One of the 10 was suitable for stock watering.

The conditions or limits in the NPDES permits consider the quality of water being discharged, the quality of receiving water, and available, practical, treatment technology. Industry is required to monitor the discharges on a periodic basis, usually daily, weekly, or monthly, and report results to EPA. The NPDES permits and related monitoring data help to estimate the quantity and quality of discharge allowed to enter the off-site environment.

Intensive studies of the influence of uranium mining on water quality and availability have not been conducted. Most investigations to date have been site-specific, of relatively short duration, and focused on the influence of surface discharge or subsurface seepage on water quality. Baseline studies from specific projects typically consist of quarterly or semi-annual sampling and are oriented toward milling instead of mining activities. The effects of dewatering on depleted water supplies or on water quality shifts in the aquifers of an area are rarely considered, and, then only on a mine-by-mine basis. Rarely are soil, stream sediment, and biologic samples collected in the preoperational period for radiologic analysis.

3.2.2 Mine Water Management

Figure 3.3 shows a scheme for considering the fate of water discharged from underground and open pit mines, including principal sources and sinks, most of which affect both water flow and quality. Broken lines in Fig. 3.3 indicate less important sequences with respect to water quality. For example, those mines that handle all water by on-site evaporation are likely to involve small volumes of water, and impacts on groundwater as a result of seepage are also likely to be small.

Mine drainage is surface water or groundwater flowing from a mine or an area affected by mining activities. Mine related point and nonpoint pollution sources can contaminate both surface water and groundwater throughout all phases of mining, that is, during mineral exploration, mine development, mineral extraction, processing, transport, and storage, and waste disposal. While mine-related point pollution sources usually include only milling and processing plant discharges and mine dewatering discharges, nonpoint sources can occur during any or all phases of mining. The chemical and physical characteristics and the mode of transfer of these nonpoint sources are variable and depend upon, among other things, the mineral being mined, its geo-

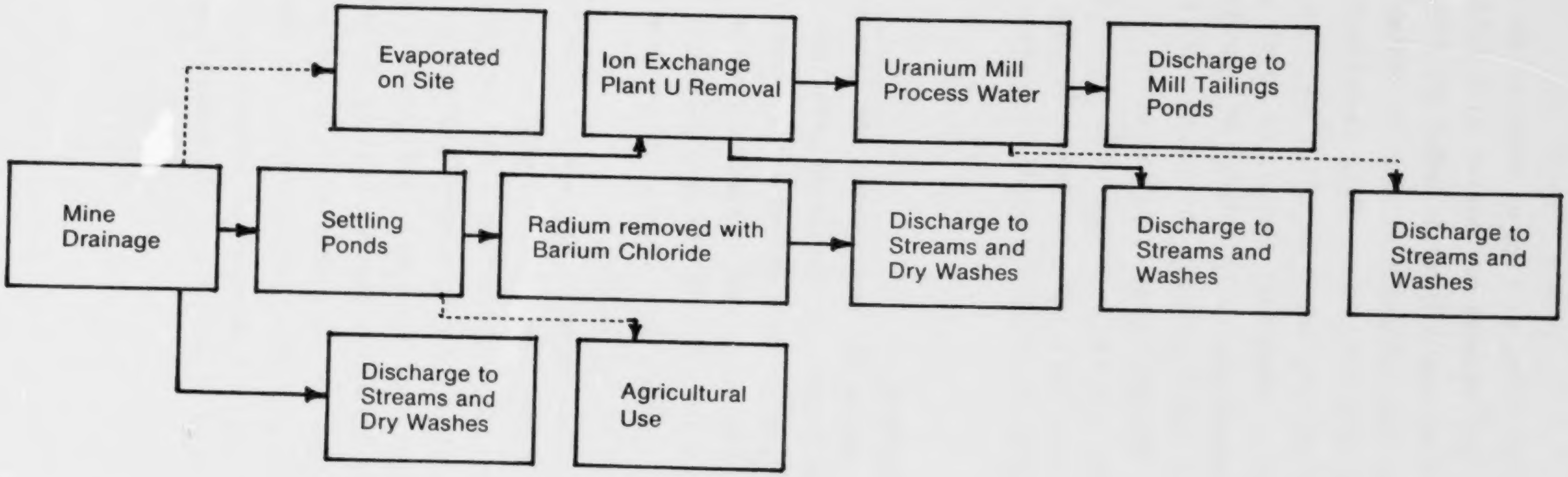
Mine Drainage
Evaporated on Site
Settling Ponds
Discharge to Streams and Dry Washes
Ion Exchange Plant U Removal
Radium removed with Barium Chloride
Agricultural Use
Uranium Mill Process Water
Discharge to Streams and Dry Washes
Discharge to Mill Tailings Ponds
Discharge to Streams and Washes
Discharge to Streams and Washes

Figure 3.3. Disposition of drainage water from active surface and underground uranium mines

logic environment, the interrelations of all associated hydrologic systems (both surface water and groundwater), and the type of processing, transportation, storage, and waste disposal methods. Some mine-related nonpoint pollution sources are as follows (EPA77a):

1. suspended solids carried by immediate surface runoff
2. dissolved solids carried by immediate surface runoff
3. suspended and dissolved solids in proximate subsurface water seepage
4. dissolved solids in groundwater recharge
5. dissolved solids in groundwater discharge
6. uncontrolled contributions from mine-related point sources:
    a. high instantaneous concentrations of regulated pollutants in excess of effluent discharge guidelines, but falling within the NPDES instantaneous and daily average discharge limitations
    b. unregulated minor contaminants in point source discharges which are not specifically included under NPDES effluent limitations
    c. untreated mine dewatering discharges during or following major storm events (NPDES point source treatment systems may be bypassed during storm events of greater than a 10-year, 24-hour intensity)
7. reclaimed mine area and undisturbed area drainage diversion discharges
8. surface water and groundwater contamination and degradation induced by mine-related hydrologic disturbances and imbalances

Typically, waters affected by mine drainage are chemically altered by an increase in iron, sulfate, acidity (or alkalinity), hardness, TDS, and various metals, and are physically altered by an increase in suspended solids such as silt and sediment (Anon69, Mi68).

Many but not all uranium mines dewater at rates of 1 to perhaps 20 m^3/min. Typically, the water from the mine goes to settling ponds and then either to the mill or a nearby stream, dry wash, river, etc. Depending on the amount of mine water recycled in the mill and the amount of water pumped from the mines, there may or may not be any release to streams or arroyos. In at least one instance in the Grants Mineral Belt, mine water is totally recycled through the mine to enhance solubilization of uranium which is removed with ion exchange columns. Large evaporation ponds and some seepage losses help maintain a water balance and minimize releases to streams, arroyos, etc. Increasing competition for water in the western states is likely to induce maximum mine water reuse (in the mill), reinjection, or use for potable supplies (Hi77) or power plant cooling.

When mill tailings ponds are used for final disposal of mine water, there is significant addition of chemical and radiochemical contaminants to the water in the course of milling. After treatment to reduce suspended solids, mine water may be recycled for use in the milling process or released to nearby streams, necessitating radium removal and reduction of suspended solids. Dissolved uranium in mine water, if present in concentrations exceeding about 3 mg/ℓ, is recovered by ion exchange columns. Settling ponds at the mines remove suspended solids. The water is then conveyed to receiving streams or to the mills for uranium recovery and (or) to satisfy mill feed water requirements. There are rather rigid requirements for release to surface water compared to groundwater.

A recent survey of 20 U.S. uranium mills (Ja79a) found large variation in the degree of water recycling. Where mine water is readily available, it probably is reused less than in water-short areas. More efficient water use by uranium mills possibly could increase the amount of (relatively) high quality mine water being discharged to the environment; lessen adverse impacts of mill tailings disposal by reducing the amount of liquid; and make mine water available as a source of potable water (after treatment) in water-short areas such as Churchrock and Gallup, New Mexico (Hi77). To date, water quality deterioration related to seepage and accidental release of tailings to surface streams has received the most study and has been the focus of regulatory programs. In the future, it is likely that water quantity issues will become increasingly important, particularly in areas where water supplies are already limited and where extensive dewatering necessarily accompanies mining.

Of 20 uranium mills surveyed, 6 reported part or all of the mill feed water came from mine drainage (Ja79a). In New Mexico, 19 of 30 mines surveyed by the State Environmental Improvement Division (J. Dudley, written communication) had off-site discharge to arroyos ranging up to 19 m^3 per minute. Those mines with no discharge utilized evaporation ponds or used the water for dust control. Most of the mines discharged to arroyos. In several instances, however, water was piped to a nearby mill at flow rates of 5 to 8 m^3 per minute. Relatively small quantities of mine water were used for sand backfill of mines, in-situ leaching of old workings, and irrigation of grasslands. In summary, New Mexico mines discharge 66 m^3 per minute off-site.

Of this, 12 m^3 per minute is routed to mills, and the balance is discharged directly off-site from the mine. Average discharge to streams and arroyos for the active underground mines in the Grants Mineral Belt on the whole was 1.8 m^3/min, whereas 12 mines in the Ambrosia Lake District averaged 1.7 m^3/min. In New Mexico, all mines discharging to an arroyo practice radium removal with approximately 90% efficiency. Uranium removal from mine water discharge occurs in all but two active mines. Future trends are likely to reflect increased discharge from the mine to the environment. Settling ponds, radium removal, or both will be used to meet discharge permit requirements.

In Wyoming, discharge from both surface and underground mines may be used as process water for uranium mills, discharged to surface streams, or used for irrigation. For example, at the North Morton underground mining operation, approximately 2 m^3 per minute of mine water discharge will be used to irrigate 800 hectares of alfalfa. At the South Morton surface mine operation, a like amount of discharge will become mill feed water.

A survey of all active U.S. uranium mills showed that 14 of 20 make no use of mine water (Ja79a). This may reflect mines where water simply is not encountered or the fact that mines and mills are not co-located. Most mills depend on deep wells, except in New Mexico where mine water is the main mill water supply. Table 3.9 summarizes water sources for U.S. uranium mills. Proposed NRC regulations on mill tailings disposal (44 Fed. Reg. 50012-59) purport to make long-term tailings isolation the primary consideration in mill siting. In areas subject to severe natural erosive or dispersive forces, this may mean that mills cannot be sited in the vicinity of mines. This may have effects on use of mine water for milling.

Table 3.9 Summary of feed water sources for active U.S. uranium mills

Water Source	No. of Mills
Rivers, Reservoirs	4
Wells	8
Springs	1
Unknown	1
Mine Water	3
Mine Water and Wells	3
	20

Source: Ja79a.

Although underground mining is now dominant in the Grants Mineral Belt, the greatest number of mines are small stripping operations that have long been inactive. This type of mining activity has apparently had little adverse impact on water resources. Few data are available on drainage assessment of large open pit mines such as the Jackpile-Paguate. The St. Anthony pit discharges about 0.076 m^3 per minute. Usually, the ore is above the water table. Any water present on the mine floor presumably is flood runoff or discharge from a nearby underground mine. Other strip mines in the Mineral Belt were not studied; hence, no conclusions were drawn (J. L. Kunkler, USGS, in preparation).

Mine dewatering is done either by pumping the mine pit/shaft directly or by drilling high capacity wells peripheral to the mine and pumping a sufficient volume of water to at least partially dewater the sediments. Because of the great volume of water that must be removed from an aquifer, the latter method is impractical for deep underground uranium mines. This is particularly true for the artesian aquifers of most of the deeper mines in the Grants Mineral Belt. More commonly this method is reserved for shaft sinking and open pit mines to depths of several hundred feet. Most underground mines are dewatered by pumping the water that collects in the mine itself. Borings ("longholes") made into the ore body for assay work and explosives facilitate drainage. There is considerable difference in the quality of water depending on the dewatering method used. Water removed from wells adjacent to the mine typically is representative of natural quality, but water removed from the mine can be high in radionuclides, stable elements, and suspended solids. In large part this is due to the disruptive nature of mining. However, more subtle, chemical processes of oxidation and bacterial action, aided by evaporation and free flow of air in the mine, are also operative.

The extent to which uranium exploration adversely impacts water resources is not well understood. Land surface disruption from drilling pads and access roads obviously affects erosion rates and results in mud pits and piles of contaminated cuttings on or near the land surface. Subsurface effects are less obvious. A potentially serious one is interaquifer connection via exploratory boreholes. In Wyoming, 6 million meters of exploratory drilling took place in 1979. Although State law requires mining companies to plug the holes after drilling, it is common practice to install only a surface plug and to rely on the drilling mud to effect a seal at depth.

Similar situations are likely in New Mexico and Texas. Shortages of funds and personnel to oversee proper completion and abandonment exist at the State level.

Hydraulic effects of water released from mines, whether from pumping or gravity flow, include increased surface discharge, recharge of shallow aquifers by infiltration, and decline of static water levels in formations intersected by the mine or related cone of depression. Of most concern are the effects relating to mine water discharge on downstream users and any influences, direct or indirect, of pumping/dewatering on water quality in the ore body and contiguous strata. In some locations, the Grants Mineral Belt, for example, the ore body is also a major regional artesian aquifer; hence, dewatering affects present water levels and will affect water levels at least to the year 2000, with complete recovery taking much longer. The extent and significance of uranium mine dewatering are as yet poorly documented. Recent studies have been made in New Mexico where dewatering is of concern because of the influence on regional groundwater availability for municipal use and in relation to return flows to the San Juan River (NRC79b; Ly79). In Wyoming, static water levels in wells on ranches adjacent to uranium mines owned by Exxon, Kerr-McGee, Rocky Mountain Energy Co., and other companies southwest of Douglas and between Pumpkin Buttes and Douglas have reportedly dropped 7 to 10 meters (Anon79). Water quality changes associated with dewatering generally are unknown and not specifically monitored regardless of the mining area location.

Water quality associated with dewatering is generally good, although suspended solids may be high, as expected. Discharge from dewatering wells will be low in suspended solids because of filtering by soil and rock aquifers. Overall water quality from dewatering wells, particularly for underground mines, is likely to be representative of ambient conditions in the ore body and, to a lesser extent, the adjacent formations that may also be dewatered.

Recent USGS work on groundwater in the San Juan Basin Region has indicated that mining expansion will have a significant impact on the water yield of the Morrison Formation (Ly79). In this study, although no water quality data are derived, the recharge and mine dewatering parameters that impact the expected drawdowns in the aquifer imply that a total of 7.03×10^8 m^3 of water will be produced by the 33 planned or announced mines by the year

2000. If the projected development of 72 mines occurs, dewatering would exceed 1.48 x 10^9 m^3. The model also estimates that flow in the San Juan River will decline very slightly (0.05 m^3/min). Similarly, flow in the Rio Grande Valley would be reduced by 0.85 m^3/min. The impacts will continue after mining and dewatering cease.

Table 3.10 summarizes New Mexico uranium mine discharge in relation to mine type, depth, and status (active or proposed). Note that projected mining is primarily underground and represents an average increase in mine depth of 275 percent and an increase in dewatering rate from 2.4 to 13.8 m^3 per minute. One would expect numerous water quality and quantity issues to arise if these projections materialize. For example, competition for water supply is likely to be widespread throughout the Upper Colorado River Basin, and uranium mines/mills are already relatively large water users. Dewatering and discharge require no water rights under State water laws in New Mexico, but the water is essentially wasted. Use of water in mills constitutes a beneficial use of water, and state water laws therefore require filing for water rights. Such filings may be denied upon protest from existing water users.

Inactive uranium mines and related wastes also influence water quality, particularly as a result of chemical and physical transport by surface water runoff. The main reasons why mine waste piles erode more quickly than undisturbed soils are lack of topsoil, steep angle of slopes, presence of toxic elements and buildup of salt in the near surface, and poor water retention characteristics. Usually, inactive surface and underground uranium mines are not a source of direct discharge of water, be it contaminated or of ambient quality, because of the low rainfall-high evaporation characteristics of the western uranium regions, static groundwater levels deep below the land surface in mining areas, and, in a few instances, recontouring of mined lands such that drainage is internal. Whether mines contaminate groundwater by groundwater leaching or by recharge contacting exposed oxidized ore bodies is poorly documented. Preliminary feasibility studies by the U.S. Geological Survey (Hi77) indicate generally good quality water from one inactive underground mine in the Churchrock area of New Mexico. It is possible that this water may be used as a municipal water supply for Gallup, New Mexico.

Table 3.10 Current and projected uranium mine discharges in the Grants Mineral Belt, New Mexico

Mine Type	Number of Mines	Average Depth (m)	Average Discharge (m^3/min)
Active			
Underground	33	248	2.42
Open pit	3	48	0.045
Proposed			
Underground	46	681	13.8
Open pit	0	N/A	N/A

Source: Environmental Improvement Division, State of New Mexico.

For most uranium regions, the volume of discharge from inactive mines to surface water bodies, though poorly documented, is believed to be less significant than that from active mines. The degree to which inactive mines contribute contaminants, directly or indirectly, to adjacent water resources can only be qualitatively assessed. The significance of inactive mines is highly dependent on regional setting and mine type.

Inactive surface mines in Texas are, with rare exception, not a source of direct discharge to surface water. It is unknown if there is any adverse impact from standing water in the mine pits, the most recent of which have been final-contoured with an internal drainage plan. Various observers suspect that water quality deteriorates when overland flow crosses mine spoils associated with overburden piles (It75 and He79). Water in the mine pits is unsuitable for potable and stock use due to high stable element contents, but it is generally acceptable in terms of radioactivity. Water in Texas open pit mines is a combination of runoff and groundwater. Before release from a mine, water is put in retention ponds to reduce total suspended solids. Holding ponds are used for storing mine water, and discharge is not allowed unless such discharge does not adversely affect the receiving water. Usually, water in the holding pond is returned to the mine for permanent impoundment. Dilution by runoff and direct rainfall and concentration by evaporation affects water quality, which the Texas Health Department (L. Cook, personal communication, 1979) reports as meeting radium standards for potable water. However, one survey of water in mine impoundments revealed that none of it would be suitable for potable or stock water use (It75). Whereas 50 to 100 pCi/ℓ of radium-226 is common in active open pit mines and concentrations of several hundred picocuries per liter may be present in groundwater within the ore body, concentrations in the ponds are between 1 and 5 pCi/ℓ after abandonment and reclamation. Over the long term, sedimentation and sorption processes in the surface impoundments are likely to improve the quality of surface water and the water that recharges the groundwater system (L. Cook, personal communication, 1979).

Harp (1978) recently presented a historical recap of water quality impacts from uranium mining and milling in the Shirley Basin area and on the Pathfinder Mine in particular. Self-monitoring data had been collected since 1959. Also available were results of one-time water quality and biological surveys by Wyoming Fish and Game Department and the Environmental Protection Agency, as well as self-monitoring data for NPDES permits, and various other reports, investigative studies, and miscellaneous reports. The study concluded that significant increases in ambient uranium and radium has occurred in nearby stream water and sediments. Initial strip mining and mill processing operations were primarily to blame, but previous acid-leach solution mining also had a decided impact. The major, continuing source of contamination to the Little Medicine Bow/Medicine River System is the mine dewatering effluent from Pathfinder Mines Corporation (a mine/mill complex) (J. Giedt, USEPA, written communication, 1980).

Little is known concerning drainage from inactive mines in Wyoming and Utah. It is likely that at least some of those in Utah are draining, analogous to the situation in Colorado. Few active surface mines in Wyoming drain to surface water. In the Front Range of Colorado, it is estimated that perhaps half of the mines discharge at least part of the year; whereas, in the Uravan Mineral Belt very few mines drain, and those that do are located in the lowland areas and (or) are deeper than the average for the area. From the North Park area southward to Durango mines occasionally drain, particularly those intersecting the Entrada Sandstone. Flow rates are highly variable, ranging from slight seepage to perhaps 8.5 to 17 m^3 per minute (D. Collins, personal communication, 1979). In the Yellow Cat area of Grand County, Utah, a westward extension of the Uravan Mineral Belt, the streams are ephemeral, and the few springs in the area drain inactive uranium mines dating back to at least 1950 (Ca64; D. Collins, personal communication, 1979).

The foregoing analysis indicates that (1) U.S. uranium mills make little use of mine water; (2) mine drainage is primarily to the environment, with occasional use for agriculture, sand backfilling of mines, construction, and potable supply; (3) a few active surface mines in Wyoming and many underground mines in New Mexico have the greatest discharge to the off-site environment; (4) inactive surface mines do not appear to adversely impact groundwater quality, although the effects of spoils piles may be a source of surface water contamination; and (5) selected inactive underground mines in Colorado and possibly adjacent portions of Utah may discharge water enriched in radionuclides and trace elements. Inactive mines may effect groundwater in New Mexico. Not enough as yet is known about oxidizing conditions, aquifer connections, etc. to reach definitive conclusions. Subsequent sections of the report quantify the volume and quality of mine discharge from existing mines in Wyoming and Texas and deep underground mines in New Mexico. In New Mexico, both the United Nuclear Corporation and Kerr-McGee Nuclear Corporation mills get all their water from mines. Three future mills will all use mine water, hence mine water is important for milling in New Mexico (B. Perkins, State of New Mexico, written communication, 1979). Analysis of the hydraulic and chemical effects of mine discharge in terms of environmental quality and public health are of considerable concern.

3.2.3 Water Quality Effects of Mine Water Discharge

3.2.3.1 Behavior of Contaminants in the Aqueous Environment

Mine drainage in general can cover a broad range of pH values. Drainage from uranium mines is usually alkaline and not considered as detrimental as acidic mine drainage, which occurs when acid producing minerals are associated with the ore body. In the western states, overburden material is often highly alkaline or saline and creates the potential for alkaline drainage.

Environmental problems associated with alkaline and saline drainages are not well documented (Hi73).

Water quality impacts from uranium mining are a function of both quality and quantity of discharge. Underground and surface mines commonly require dewatering prior to or during the ore removal phase, although there is considerable variation from one area or mine to another. In New Mexico, large and medium sized mines are essentially dry, whereas mines in Texas and Wyoming require extensive dewatering. Regardless of location, underground mines rarely are dry and many require extensive dewatering. Considering the variety of water management measures, regional differences in contaminants and receiving waters, and geochemical characteristics of ore bodies, detailed discussion of the effects of mine drainage or mining, in general, on water quality must await further site - or area - specific study. It is questionable if sufficient data on mine drainage exist to assess effects on biota and the fate of contaminants in surface or sub-surface water bodies.

Limited data from Colorado, Texas, Wyoming, and New Mexico suggest adverse impacts on water quality from discharge of mine water. Effects of dewatering on deep groundwater quality are very poorly documented; hence, no conclusion as to relative significance is drawn. With respect to surface water resources, discharge of mine water and overland movement of water and suspended or dissolved contaminants may be significant. Because of the dearth of data on overland flow, emphasis herein is on contaminants discharged via mine drainage water. Substantial studies to evaluate sediment yield and quality from lands mined for uranium, particularly from areas of surface mining, have not been conducted, although recent work in Texas (He79) is a notable exception.

Elements such as uranium, radium, molybdenum, selenium, zinc, and vanadium may be enriched in point and nonpoint discharges from uranium mines. The dispersal, mobility, and uptake of such elements are directly relevant to the subject of this report. We reviewed selected literature and field data to at least qualitatively understand what processes and elements are most significant and to thereby strengthen some of the underlying source term assumptions in the transport and health effects modeling. Despite the annual chemical load introduced to ephemeral streams by both dissolved and suspended constituents in mine effluent and overland flow from mined lands, waste piles etc., a number of processes affect the concentrations in the ambient environment. These include dilution, suspended sediment transport, sorption and desorption, precipitation, ion exchange, biological assimilation or degradation, and complexation.

3.2.3.1.1 Dilution and Suspended Sediment Transport

In many uranium mining regions there is flooding or flash flooding. Such storms may well be the only runoff event for a year or more at a time. It is worthwhile then to consider some of the effects of such events on mobilization of contaminants associated with uranium mining wastes. Typically there is significant interdependence between the physical and chemical processes.

A principal physical process is dilution. This will reduce concentrations of pollutants released to surface waters, but is considered to play a relatively minor role over the short term for water percolating through the soil to sources of groundwater in arid or semiarid regions.

Transport of suspended sediments in floods is another dominant process. Suspended load is largely a result of physical, hydraulic processes, hence elements that are rapidly and thoroughly removed from solution as a result of solubility limits, precipitation with other ions, ion exchange, and sorption may well be transported in the suspended load. In metal mining areas of central Colorado, total and dissolved metal loads in streams are greater during high flow periods, apparently a result of flushing from mines and tailings piles and scouring of chemical precipitates from stream substrates (Mo74). Typically, total and dissolved loads decrease downstream, regardless of discharge. Increase in iron in the downstream direction reflects scouring of precipitate--an amorphous, hydrated ferric oxide. Dispersal occurs for quite a distance downstream.

3.2.3.1.2 Sorption and Desorption

Sorption can play an extremely important role in purifying waters, particularly if infiltration or percolation is involved. This is especially true when contaminant concentrations are too low to undergo precipitation reactions. Virtually every ionic species will be sorbed and removed to some extent except for chloride and, to a lesser extent, sulfate and nitrate. These seem to pass through soils and alluvium without significant sorption (Ru76). Sorption processes can be highly specific, depending on the type of contaminant and the physical and chemical properties of both the solution and the porous medium.

There have been numerous laboratory studies on the sorption, leachability, and mobility of stable elements in various types of soil. A recent review of the literature on radionuclide interactions in soils specifically discusses radium, thorium, and uranium, which are especially pertinent to uranium mining (Am78). Distribution coefficients, the ratio of concentration in soil to that in water, ranged between 16 and 270 for uranium in various soil-river water systems, between 200 and 470 for radium, and on the order of 10^5 for thorium at pH6. These observations appear consistent with the generally accepted ideas that uranium is relatively mobile, thorium extremely immobile, and radium somewhere in between in natural water systems (NRC79b; Ku79; Ga77a).

Adsorption is believed to be important for cadmium, copper, lead, and nickel, insofar as these are transported in the suspended fraction, whereas manganese and zinc are primarily in the dissolved fraction. Adsorption of metals onto precipitated manganese oxides or hydroxides at elevated pH is probably insignificant in the case of most uranium mine discharges insofar as these are alkaline and, furthermore, discharge to or co-mingle with other streams that are alkaline.

Radium sorption and desorption tests done on uranium mill waste solids and river sediments collected from several locations in the Colorado Plateau (Sh64) and in Czechoslovakia (Ha68) showed that leaching is primarily controlled by the liquid-to-solid ratio, i.e., the volume of leaching liquid per unit weight of suspended solids. Natural leaching from mining and milling waste solids freely introduced to rivers in the past is one of the major factors in radium contamination of rivers (Ru58). Although settling ponds are now used to remove or at least reduce suspended solids from active mine discharges, the dissolved radium load sorbed on sediments presents a source term that may be somewhat analogous to river sediments contaminated by dissolved and suspended milling and mining wastes. The manner in which the radium is mobilized and the significance is poorly understood and bears further investigation. Apparently there is a "leap frog" transport mechanism involving combined chemical and physical weathering processes. There should be a marked downstream attenuation of both dissolved and sorbed/precipitated radium inventories insofar as sediment burial and dilution take place and leachability limits are reached, i.e., no more radium can be removed regardless of the duration, frequency, or intensity of agitation. Should the stream eventually discharge into a reservoir, it is unlikely that renewed leaching will take place. Shearer and Lee (Sh64) did not account for some factors that may be locally significant, such as bio-uptake along the stream/river, use of water for irrigation, number of uranium facilities discharging, and other local factors.

Experiments were conducted in Japan (Ya73) to determine uranium adsorption and desorption using carbonate solutions and three soil types (alluvial, sandy, volcanic ash). Very high adsorption ratios and very low desorption ratios of uranium characterized the various soil types in contact with stream water and help explain the decrease in soluble uranium with flow distance from mines (Ma69). When wastewater flows into streams at the maximum permissible concentration (1.8 mg U/ℓ) recommended (ICRP64), Yamamoto et al. (Ya73) conclude that the uranium behaves as a uranyl carbonate complex anion and that essentially complete sorption readily occurs in the presence of (Japanese river) water which contains 15 to 39.9 mg/ℓ bicarbonate. Since this is similar to concentrations in surface waters of uranium regions in the western states, similar results are expected.

Sorption or desorption of heavy metals such as Co, Ni, Cu, and Zn in soils and fresh water sediments occurs in response to the aqueous concentration of metal, aqueous concentration of other metals, pH, and amount and strength of organic chelates and inorganic complex ion formers in solution (Je68). Other controls on the heavy metal concentrations in soil and fresh water include organic matter, clays, carbonates, and oxide and hydroxide precipitates.

To what degree solubility acts as a limit on stable element concentrations in natural waters is unclear. The crystallographic form or even the chemical composition of a precipitate are often unknown. Elements such as iron, aluminum, manganese, and titanium form insoluble hydroxides and are likely to exceed equilibrium solubility limits (An73). Hem (He60) partially disagrees, saying "it is not unreasonable to assume equilibirum for the iron species in water." Whether mine discharges or overland flow from mined areas are in equilibrium is unknown, but it is doubtful considering the underground or flash-flood origin of such waters. The non-equilibrium aspects of certain peak runoff events has been documented for major streams of the world (Durum and Haffty, 1963). Metals such as iron, aluminum, manganese, and titanium, which readily form rather insoluble hydroxides as particulates or colloids, may be dissolved from suspended minerals during high flow conditions. Organics present in such flood waters may assist through formation of soluble complexes. Resulting metal concentrations may be higher than solubility and redox relationships alone would indicate.

3.2.3.1.3. Precipitation

Probably one of the most significant processes affecting stable element solubility in natural water systems is adsorption on hydrous ferric and manganese oxides. Jenner (Je68) believes this is the principal control on the fixation of Co, Ni, Cu, and Zn (heavy metals) in soils and fresh water sediments. For example, ferric hydroxide adsorbs one to two orders of magnitude more SeO_3 per unit weight than clays, and 90 to 99 percent adsorption is possible at a pH of seven to eight typical of most western streams (Ho72). At neutral or slightly alkaline pH, both iron and manganese are poorly soluble in oxidizing systems and, in general, exhibit very similar chemical behavior, although manganese is slightly more soluble. Fixation of selenium in soils, particularly by iron oxide or as ferric selenite, renders it unavailable to agricultural and forage crops, although specific selenium-accumulating plants can remove the element and, upon decomposition, release it in water soluble forms, such as selenate and organic selenium compounds, available to other plants (Ro64). The behavior and mechanism of selenium adsorption (as selenium oxyanion) by hydrous ferric oxides is readily extended to the interpretation of other similarly bound minor elements (Ho72). Mobile selenium oxyanion in slightly alkaline waters might be carried to streams by surface runoff or in groundwater. Selenite selenium sorbed upon ferric hydroxide should be transported in surface waters at neutral or slightly acid pH. Other metals forming highly insoluble hydroxides in the pH range of 6 to 9 include copper (above pH 6.5), zinc (above pH 7.5), and nickel (above pH 9). Molybdenum is thought to hydrolyze to the bimolybdate ion under acid conditions and precipitate with iron and aluminum. Aerobic or oxidizing conditions in the vadose zone are favorable for the development of many of these oxides (Cu, Fe, Mn, Hg, Ni, Zn, Pb). Reducing conditions deep in saturated zones generally lead to increased mobility of these metals.

Reducing conditions that can exist in the presence of organic material (bituminous or lower ranking coals, anaerobic bacteria, fluidized humates) can lead to precipitation reactions favorable for removing contaminants from mine waters. Reduction of uranium to the quadrivalent state and its fixation on clays would play the major role in protecting groundwater supplies from uranium if the appropriate reducing agents were present in soils (Ga77a; Ku79). Inorganic reducing agents could include ferrous iron and hydrogen sulfide produced by the action of an aerobic bacteria on sulfates. Natural reducing conditions can also, theoretically, cause the formation of such native elements as arsenic, copper, mercury, selenium, silver, and lead, which are all quite insoluble in their elemental form (Ru76). Hydrogen sulfide or other sulfides, if available, will serve to reduce the concentrations of such metals as arsenic, cadmium, copper, iron, lead, mercury, molybdenum, nickel, silver, and lead.

The metal-scavenging of hydrated iron oxide precipitate has been documented in a mined area of Colorado where relatively acid schists and gneisses give rise to acid runoff that dissolves large quantities of aluminum, magnesium, and zinc. Runoff from a nearby drainage basin underlain by basic rocks containing base and precious metal veins carries considerably less metal. However, manganese oxide precipitated with iron oxide contains large quantities of metals. Ferric hydroxide precipitates from aerated water solutions containing more than 0.01 ppm iron at pH values of 4.5 and above, aluminum hydroxide precipitates in the pH range of 5 to 7, and manganese hydroxide precipitates above pH8 (He60; Ch54). Considering the alkaline pH of most uranium mine discharges and overland flow from non-point sources such as mine waste piles, precipitation of iron and possibly manganese seems certain. The scavenging effect of iron hydroxide at neutral to alkaline pH is considerably less than that of manganese hydroxide precipitate.

The extensive studies of mine drainage in Colorado by Morgan and Wentz (1974) revealed the effects of solubility on stable element transport. In the downstream direction, dilution and neutralization of the acid mine drainage by bicarbonate caused dissolved metal to decrease due to dilution, chemical precipitation, and probably adsorption onto ferric hydroxide precipitate. The latter creates a coating on the stream substrate for a considerable distance during low flow periods. Subsequently, flood events scour and transport the precipitates. Manganese and zinc remain primarily in the dissolved phase for a considerable distance, whereas cadmium, copper, iron, lead, and nickel concentrate in the suspended fraction and, when turbulence decreases, precipitate. The mobility sequence for the metals studied in Colorado generally follows the order $\mathrm{Mn} \cong \mathrm{Zn} > \mathrm{Cu} > \mathrm{Cd} > \mathrm{Fe} > \mathrm{Ni} > \mathrm{Pb}$. Ferric hydroxide precipitation and scavenging seems to be more important at neutral than at acidic pH's (Je68).

3.2.3.1.4 Biological Assimilation and Degradation

Biological uptake and the role it has on stable element concentrations in water is not predictively understood (An73). Plant uptake of stable elements and resulting phytotoxicity is not merely a function of how much is present in the soils or water. In the case of arsenic, the chemical form of arsenic appears more important than the total soil arsenic (Wo71). For example, water-soluble arsenic in soil created more phytotoxic effects than those with no detectable water-soluble arsenic. Soils high in reactive aluminum remained less phytotoxic, despite heavy applications of arsenic, than soils with low reactive aluminum. Selenium in soils can be present as elemental selenium, selenates, pyritic selenium, ferric selenites, and organic selenium compounds of unknown composition. Selenates and organic compounds are most available to plants, although slow hydrolysis of the other forms can occur such that they become available for plant uptake. The importance of water soluble selenium versus total selenium as the major factor affecting plant uptake has been demonstrated (La72; Gr67). Where sufficient selenium is present in plant-available form, all species will take it up in sufficient amounts to be harmful to animals (La72). Naturally occurring soils containing such available forms are geographically confined to semiarid regions or areas of impeded drainage. Such soils are not hazardous to humans and only locally are they a threat to animals.

Despite numerous examples of high selenium (up to 2.7 ppm) in surface water, particularly that associated with drainage from seleniferous soils in agricultural areas, Rosenfeld and Beath (Ro64) reported only a few cases of water-related selenosis in man or livestock. Water high in selenium is typically unpalatable to livestock and certainly to man. Lakin (La72) concluded that environmental contamination due to selenium is increasing, but hazardous concentrations are unlikely; mining and industrial wastes may cause local problems; and the effect of added selenium in waters in combination with other contaminants bears further study.

Uranium uptake by several species of native plants in the southeastern

Utah portion of the Colorado Plateau varied, sometimes strikingly, with the species, time of year, part of plant, availability of uranium in the soil, and chemical composition of the underlying rocks (Ca57). The type of rooting system and the soil moisture conditions also were influential. In some cases, there was no consistent relationship between the amount of uranium in the soil versus that in the plant ash. Plants are much less selective with respect to cadmium uptake, and it has been conclusively demonstrated that plants absorb cadmium from cadmium containing solutions and soils (Pa73; Fu73). Phytotoxic effects vary considerably with plant species. Cadmium and zinc sulfides tend to concentrate in the organic matter of soils. Upon oxidation to sulfate, plant availability increases along with solubility. Under alkaline conditions (pH8), cadmium is taken up rapidly by biota and by sediments. However, modeling of cadmium transport and its deposition in aquatic systems is very complex and encompasses many variables, most important of which are pH, carbonate content, chemical form, and competing ions.

3.2.3.1.5 Complexation

Published data on Gibbs free energies, enthalpies, and entropies of 42 dissolved uranium species and 30 uranium-bearing solid phases were recently reviewed (La78). Uranium in natural waters is usually complexed with carbonate, hydroxide, phosphate, fluoride, sulfate, and perhaps silicate. Such complexes greatly increase the solubility of uranium minerals and increase uranium mobility in groundwater and surface water. In waters with typical concentration of chloride, fluoride, phosphate, and sulfate, intermediate Eh's, neutral to alkaline pH's, and the presence of phosphate or carbonate, uranyl phosphate or carbonate complexes form and increase mineral solubility by several orders of magnitude. Sorption of the uranyl minerals carnotite, tyuyamunite, autunite, potassium autunite, and uranophane onto natural materials is greatest in the pH range of 5 to 8.5. Uranium content of small streams, in particular, can exhibit wide spatial and temporal variations due to pH and oxidation state of the water, concentrations of complex-forming species such as carbonate or sulfate, and presence of highly sorptive materials such as organic matter, certain metallic hydroxides, and clays (La78). Whereas sorption is probably a dominant control on stable element concentrations in low temperature aqueous conditions, there is insufficient information concerning specific sorbents to allow accurate prediction.

3.2.3.2 Results of Field Studies in Uranium Mining Areas

3.2.3.2.1 Colorado

Extensive studies of the effect of mine drainage on stream water quality and biota were done in central Colorado (Mo74). Although uranium was mined in 14 of the 25 areas studied, other metals were the principal products. Most of the ores were high in iron sulfides, and associated drainage was acidic. Also studied, but less intensely, was the Uravan district of western Colorado where the principal products are uranium and vanadium from Mesozoic sandstone. The Uravan Mineral Belt is different in terms of principal product and geologic features from other mining areas studied in Colorado. For these other areas, the drainage is acidic and heavily enriched in heavy metals and, therefore, somewhat atypical of most Colorado uranium mines in the Uravan area.

After a preliminary field survey of the temperature, specific conductivity, pH, stream-bottom conditions, and aquatic biota at 995 stream sites, 192 were chosen for detailed sampling and analysis during 1971-1972. The data indicate the contamination of approximately 711 kilometers of streams in 25 different areas, mostly in the Colorado Mineral Belt. The water quality effects in these areas arise from many varied causes, including active and inactive mine drainage, tailings pond seepage, drainage tunnels, and milling operations. The length of the streams affected is not absolute as it varies with the time of the year and flow conditions (Mo74).

The general findings indicate that Mn, Se, and SO_4 concentrations, and specific conductivity are poor indicators of mine drainage as natural sources can cause high values for these parameters even in undisturbed areas. Uranium mines make at least some contribution to problems of contaminated streams in central Colorado. In central Colorado, the exact impact of uranium mining on stream water quality is unknown but believed to be less important or significant in most areas as compared to impact from other mining, with the possible exception of the Boulder-Jamestown area (J. Goettl and D. Anderson, Colorado Game, Fish, and Parks Division and Water Pollution Control Commission, respectively, personal communication). Cadmium, As, and Pb exceed the U.S. Public Health Service toxicity limits, respectively, 12.5 percent, 1.4 percent, and 2.1 percent of the time. Mercury and Ag limits were never exceeded, and Cr was never detected. Iron and Mn standards were frequently exceeded by large percentages, however, these limits are only based on aesthetics. Concerning the negative impacts of the various constituents, Cu and Zn (exceeding the limits 7.8 and 9.0 percent of the time, respectively) pose the greatest threat to resident aquatic life. Mining operations in the Uravan area are a relatively minor source of metals for the San Miguel River (Mo74). Potential problem areas are settling ponds and tailings piles associated with the mining operation. Although not a source of acid drainage, these sources did cause increased concentrations of copper, iron, manganese, nickel, vanadium, and zinc in the river. Only manganese exceeded the standard for drinking water, and no metal concentrations exceeded the biological criteria. Seepage (0.003 m^3/s, pH 6.8, 3300 mg/ℓ HCO_3) from a mine tailings area into Atkinson Creek, a tributary of the San Miguel River, observed in December 1972 caused no adverse impacts.

Because of its size, proximity to population, and effects on surface water quality, extensive surface water quality investigations to assess the impacts of mine water discharge from the Schwarzwalder mine have been made (EPA72). Grab samples of the mine effluent taken in 1972 revealed 15 mg/ℓ uranium and 80 pCi/ℓ radium-226. As of 1972, overflow and seepage from the settling ponds used to treat the mine effluent significantly degrade the radiochemical quality of nearby Ralston Creek. This was confirmed by both EPA and the State/Denver Water Boards monitoring program. With 20-fold dilution, Ralston Creek downstream of the mine contained 3 pCi/ℓ and 82 μg/ℓ dissolved radium-226 and uranium, respectively. With no dilution, as during July, concentrations were 81 pCi/ℓ and 20,300 μg/ℓ. Influx of contaminated stream water to nearby Long Lake raised dissolved radium-226 to 0.8 pCi/ℓ (4-fold increase over background) and uranium to 230 μg/ℓ (20 times background). From these data, conclusions were reached that the mine water caused a 5 percent increase in the radiation dose to consumers in a local water system (based on FRC and NCRP standards and daily consumption of 1.0 liter water). If the 4.5 mg/ℓ uranium limit proposed by ICRP was used, the estimated dose increases to nearly 40 percent of the dose limit for a population group. Since 1972, the effluent has been treated for radium-226 and uranium removal. Trace metals analysis of water samples collected from the creek and the water treatment plants revealed concentrations comparable to or greater than those in the effluent as of July 20, 1978. Concentrations (μg/ℓ) were as follows:

	As	F*	Pb	Se	Zn
Mine Effluent	5	1	15	<2	18
Ralston Creek (avg)	5	1.3	32	<2	56
Water treatment plants (avg)	5	0.55	97	2	146

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*mg/ℓ

3.2.3.2.2 Wyoming

We assessed the effects of mine drainage by literature review and a limited field study in the Spring of 1979. Results of the latter conclude this section of the report. The effects of mine dewatering, in situ leaching, and mill tailings seepage on surface water quality in the Shirley Basin were previously studied by the Wyoming Department of Environmental Quality (Ha78). Sixteen years of data on aqueous radium and uranium indicated significant amounts of radium-226 and uranium reached streams because of inadequate mine dewater treatment, mill tailings pond seepage, and improper operation of a precipitation treatment unit. Uranium concentrations in stream water increased 60-fold because of mine water discharge and possible tailings pond seepage. The effects of past loadings of uranium and radium on fish propagation or migration are not clear, although biologic uptake of uranium and radium has occurred. Phytoplankton, algae, and bottom fauna organisms also do not appear to have been adversely affected, but no studies have been conducted since 1962. Long-term effects of increased radioactivity levels are known and merit further study: "There exists real need for additional studies to determine the mechanisms involved in the dispersion and ultimate disposal of uranium loaded into the drainage basin...Only after additional studies have been completed, may we understand the total and long range impact that the company's activities have had on the aqueous environment" (Ha78). This latter finding related specifically to the current (1978) loading of uranium from treated mine discharge.

Previous studies by the State of Wyoming (Ha78) found that solution mining by the Pathfinder Uranium Company noticeably affected ambient uranium concentrations in the study areas. A 1968 survey by the Department of Environmental Quality (Ha78) indicated relatively high loadings of soluble uranium and radium on stream sediments near the mine dewatering outfall. Analysis of fish skeletons indicated radium uptake corresponding to dissolved radium-226 concentration exceeding 1 pCi/ℓ. Resampling in 1970 showed a decrease in radioactivity values in sediment but a tenfold increase in fish uptake of uranium relative to other fish populations in the basin. Radioactivity concentrations in fish tissue were highest near the mine effluent outfalls but did not constitute a major source of radioactive intake by consumers.

In June 1971, the EPA Radiological Activities Section of Region VIII (Denver) made a field reconnaissance of uranium mining and milling activities in the Shirley Basin area. Radiological analyses of water and sediment samples in the Shirley Basin and in the Bates Hole drainage basin to the north unquestionably indicated significant increases in radioactivity levels in water, sediment, and fish because of effluent discharge from mines and mill tailings. Concentrations of dissolved radium-226 and uranium in mine effluent were well above background. The discharges were not considered a source of radiation dose to the populace (residents and transients) because of remoteness and lack of water use, but toxic effects on fish were of concern (M. Lammering, written communications, 1979). Monitoring in 1972 by the Wyoming Game and Fish Department showed water quality effects as far as seven miles downstream. From 1970 to 1972, radium-226 concentrations remained stable, but uranium increased. Fish samples collected in 1972 showed increased amounts of radium in the flesh compared to the 1970 results. Soil samples from a creek that received mine effluent indicated relatively large transport and enrichment of uranium and radium. Radium in particular was enriched in the sediments and showed temporal variations indicative of successive scouring and removal, presumably in flood flows. Precipitation of uranium compounds was not apparent, probably because oxidized uranyl species are quite soluble in natural water.

To further our understanding of the role aqueous pathways play in contaminant dispersal, we monitored stable and radioactive trace elements in the Spring of 1979 in surface runoff from ore, sub-ore, and overburden piles from the Morton Ranch area of active mining in Wyoming. EPA personnel selected the basic study areas and assisted in sample collecting. Most of the sampling, analysis, and interpretation was done under contract with Battelle, Pacific Northwest Laboratories and is the subject of a draft report (Wo79). Appendix G contains a more complete discussion and listing of the data. Since runoff samples were unavailable, the sampling program emphasized the collection of soil samples in well-defined runoff gullies originating at surface stockpiles of minerals. Where the drainage systems intersected flowing water, upstream and downstream samples were collected. Most of the samples consisted of the top five centimeters of soil in the bottom of the drainage channels and three core profiles. Samples of the source material were also collected.

Trace element and radionuclide analyses of runoff from the Morton Ranch area are primarily based on surface sediments and vertical sediment profiles from the dry stream beds, since few streams or other forms of runoff were encountered. Figure 3.4 shows the waterways surrounding the inactive 1601 pit area and the semi-active 1704 pit area.

Three vertical soil profile samples were collected, two in an erosional drainage area from the ore and waste pits south of the 1601 pit and one in an erosional drainage bed on the east side of the waste pile of the 1704 pit. The radionuclide and chemical constituents of these samples along with analyses of other soil samples are reported in Appendix G. The results indicate aqueous leaching based on the radium/uranium ratios of about 12 in redeposited material in the alluvial fan area of the drainage, compared to a corresponding ratio of 0.9 in the undisturbed (sub-surface) material in the top alluvium profile.

Trace element data also indicate limited transport of mine contamination with respect to uranium, and to a lesser extent, Se and V. The profile samples containing 140 and 21 ppm U resulted from material transported from the adjacent ore piles. The aqueous samples similarly showed no unusual characteristics indicative of mine wastes.

An aqueous sample and the soil profiles collected near the 1704 pit similarly showed no evidence of mine-related pollution or leaching of uranium. This observation is based on the water and particulate analyses and the radionuclide analyses. These samples constitute a worst case, since the sediment samples were collected in the redeposited material of the waste pile drainage and should show greater levels of pollutants there than would fin-

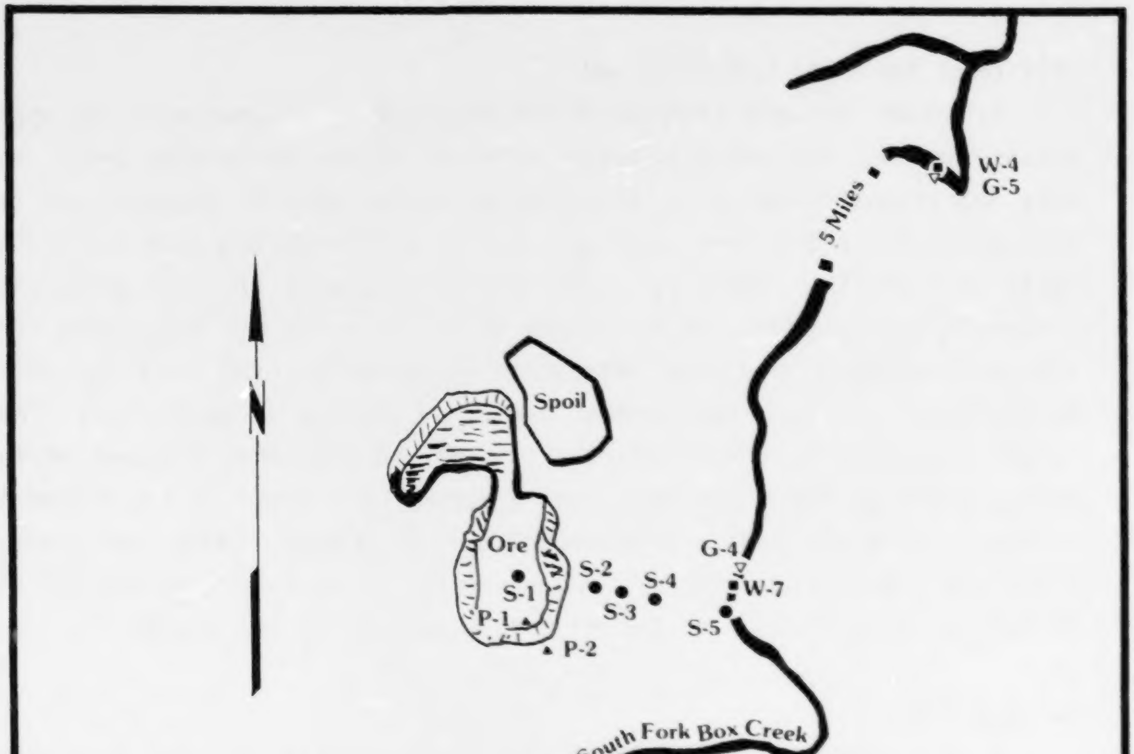


Figure 3.4 Location of mines, ore and waste storage areas and monitoring stations at the Morton Ranch mine, South Powder River Basin, Wyoming.

ally reach the South Fork Creek bed.

Pollutant releases from the Morton Ranch, Wyoming uranium mining operations were not observable in water drainages of the surrounding area. The only significant movement of mine-related wastes was the transport of the stockpiled ore in erosional drainage areas on and immediately adjacent to the waste pile of the 1601 pit. Long-distance transport of these pollutants (primarily uranium) into the South Fork of Box Creek was not observable. The strongest evidence that mine wastes are a source of local soil and water contamination is the radiochemical data, and uranium in particular. Pollutant transport is almost entirely confined to the immediate area of the mines, although there has been some dispersal via water in the ephemeral streams. There is considerable disequilibrium between radium and uranium which may indicate leaching and remobilization of uranium. The possibility of natural disequilibrium in the ore body should not be overlooked.

3.2.3.2.3 Texas

A very comprehensive field and literature survey of elements associated with uranium deposits in south Texas (He79) revealed high to very high concentrations of molybdenum, arsenic, and selenium in areas of shallow mineralization; drainages adjacent to older, abandoned mines; and in some reclaimed areas. Areas of shallow mineralization have concentrations of several tens of ppm molybdenum and arsenic and up to 14 ppm selenium. Near surface material exposed by mining may have several hundred ppm molybdenum and arsenic. Waterborne transport of suspended or dissolved solids away from open pit mines resulted from mine water discharge and (or) surface runoff and erosion of abandoned spoil piles. Molybdenum from the mining areas could potentially aggravate natural soil problems leading to molybdenosis (Kab79). Additional careful study is suggested, particularly of areas receiving mine drainage as pumped water or overland flow.

Lakes or ponds associated with 10 mine locations in Karnes and Live Oak counties contained water unsuitable for drinking without prior treatment (It75). Generally, mineralization was also excessive and rendered the water unfit for irrigation. Air and terrestrial sampling revealed no health hazards from mining wastes and mined lands, but insects and other bottom fauna in the lakes concentrated radium-226 400 to 800 times the water concentration based on dry weight of the organisms.

3.2.3.2.4 New Mexico

The principal investigations of the influence of uranium mining on water quality includes EPA and contracted (Wo79) work by the U.S. Environmental Protection Agency, the Department of Interior (Ku79), and ongoing studies by the State of New Mexico (J. Dudley, oral communication, 1979). Because of the co-location of mining and milling facilities, it is difficult to identify impacts from one versus the other.

Survey of groundwater and surface water quality in close proximity to the Jackpile-Paguate, Ambrosia Lake, and Churchrock mining areas (EPA75) revealed extensive discharges of mine water to the ambient environment, use of unlined ponds for settling suspended solids from mine dewatering, use of contaminated mine water as a potable supply (one facility), and failure of all facilities discharging to streams to have a valid NPDES permit. The volume of mine discharge, particularly in the Churchrock area and from a mine near Mount Taylor, led to use of the water for irrigation and stock. In other areas of Ambrosia Lake and near Churchrock, infiltration of mine water mixed with seepage from mill tailings ponds is causing local contamination of shallow, potable aquifers, but the problem is not considered serious and ongoing studies are underway. The State of New Mexico has installed a monitoring well network to determine temporal and spatial trends in groundwater quality. The U.S. Geological Survey, in particular, is monitoring surface flows and water quality in the Ambrosia Lake and Churchrock areas.

As part of the San Juan Basin Regional Uranium Study, the Department of Interior (DOI79) assisted by the U.S. Geological Survey (Ku79) examined selected water quality impacts from mining and milling and concluded that much of the mine effluent is suitable for irrigation, stock, and industrial use. Locally, it supports aquatic life and wildlife. Additional data on stream sediments are needed to evaluate the impact on water resources of erosion of waste rock from mines and mill tailings. It is preliminarily suggested (Ku79) that such erosion may be difficult to detect at distances of more than a few miles from the source because of the large amount of (natural) regional soil erosion. The results of the study are presently in draft form and may therefore be revised.

As part of the present study on uranium mining wastes, two New Mexico areas containing inactive mines were surveyed in the Spring of 1979. Stable and radioactive trace elements were monitored in surface runoff from sub-ore and overburden piles in Ambrosia Lake and in the nearby Poison Canyon area. EPA staff selected the basic study areas and assisted in sample collection. The bulk of the sampling, analytical, and interpretation phases was done by Battelle, Pacific Northwest Laboratories (Wo79). Appendix G contains the data and discussion. As in the case of the Wyoming study area, the sampling program emphasized stream sediment sampling, cores, and shallow (5 cm thick) grab samples at the land surface. Samples of the source material were also collected. Figures 3.5 and 3.6 show the location of the study areas and sampling stations in New Mexico. Samples of the source material were collected at one of the two New Mexico drainage systems investigated. One system in Poison Canyon, New Mexico, was adjacent to several small surface operations as well as an underground mine site. For this system, no single source could be defined for the runoff constituents.

The Poison Canyon mine drainage system is a dry creek bed. The course of this creek passes an abandoned underground mine site from which it can receive runoff water. It then passes through a dirt roadway and follows a course adjacent to some small open pit mines. After a distance of several hundred kilometers, it joins a second branch drainage that originates next to a waste pile from one of the open pit mines. Samples were collected along this waterway starting with a background sample upstream of the underground mine about 200 m from the road. The first downstream sample was collected about 130 m downstream from the roadway. This was upstream of the runoff source originating in the open pit mine. The remaining samples were collected along the drainage way (Fig. 3.5), below contamination sources from the open pit operations.

A second site, the San Mateo Mine and environs, is located in the southeast portion of the Ambrosia Lake mining district. Large mine waste piles, a heap leaching operation, and a mine drainage pond are prominent at the site, which drains northward to San Mateo Creek.

Soil composites were collected at the waste pile and heap leach pile. These represent the source term for possible contamination of the watershed. The drainage samples were collected following one channel down the waste pile face to the intersection with San Mateo Creek, which was followed for a distance of 500 to 600 m from the site. Additional samples were collected in the gullies leading from the heap leach area and one of the off-site gullies. The latter represents blank soil upstream of the drainage water. Sampling sites are noted in Fig. 3.6. No significant contamination from the under-

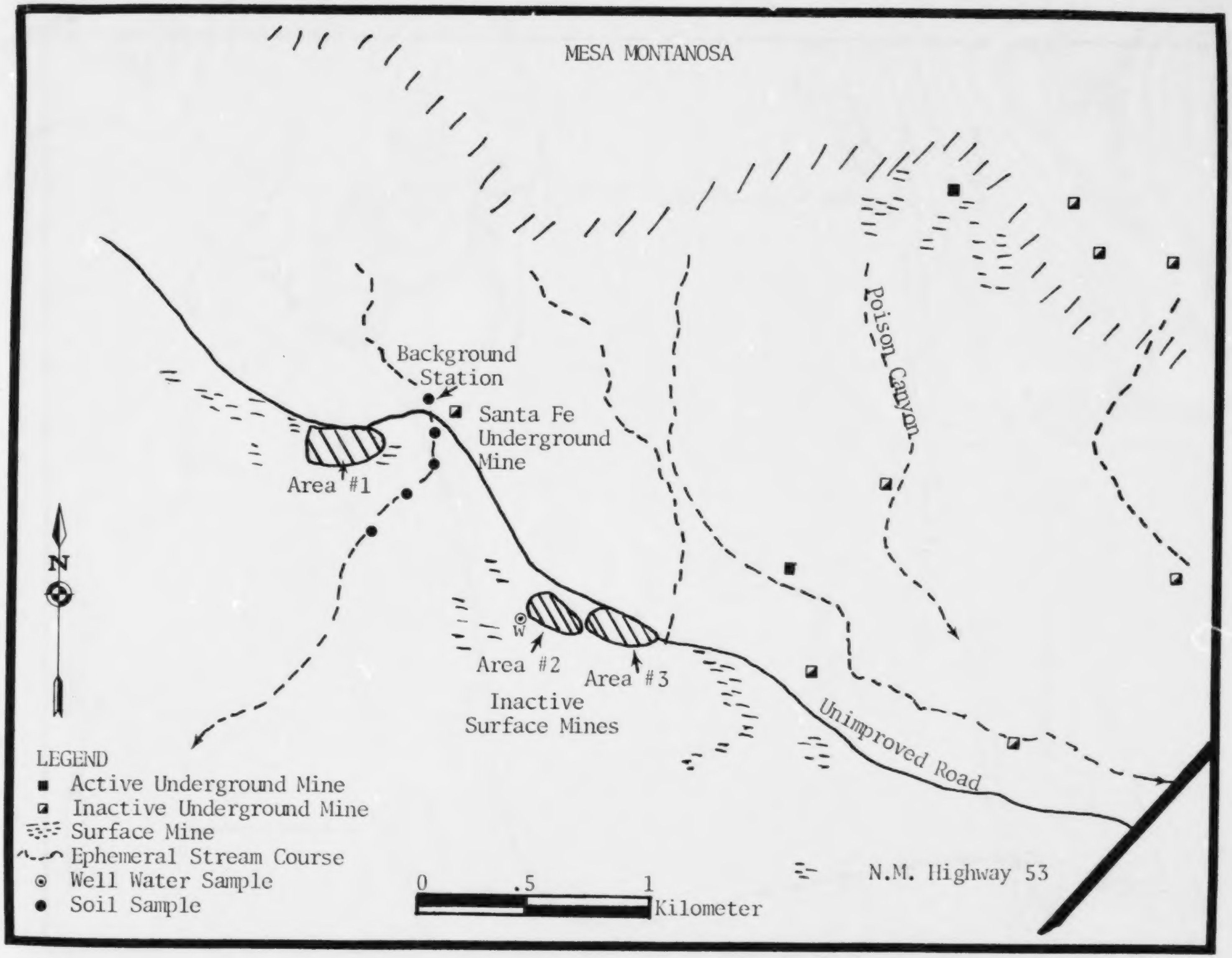


Figure 3.5 Location of study areas, sampling stations and uranium mines, Poison Canyon area, McKinley County, New Mexico.

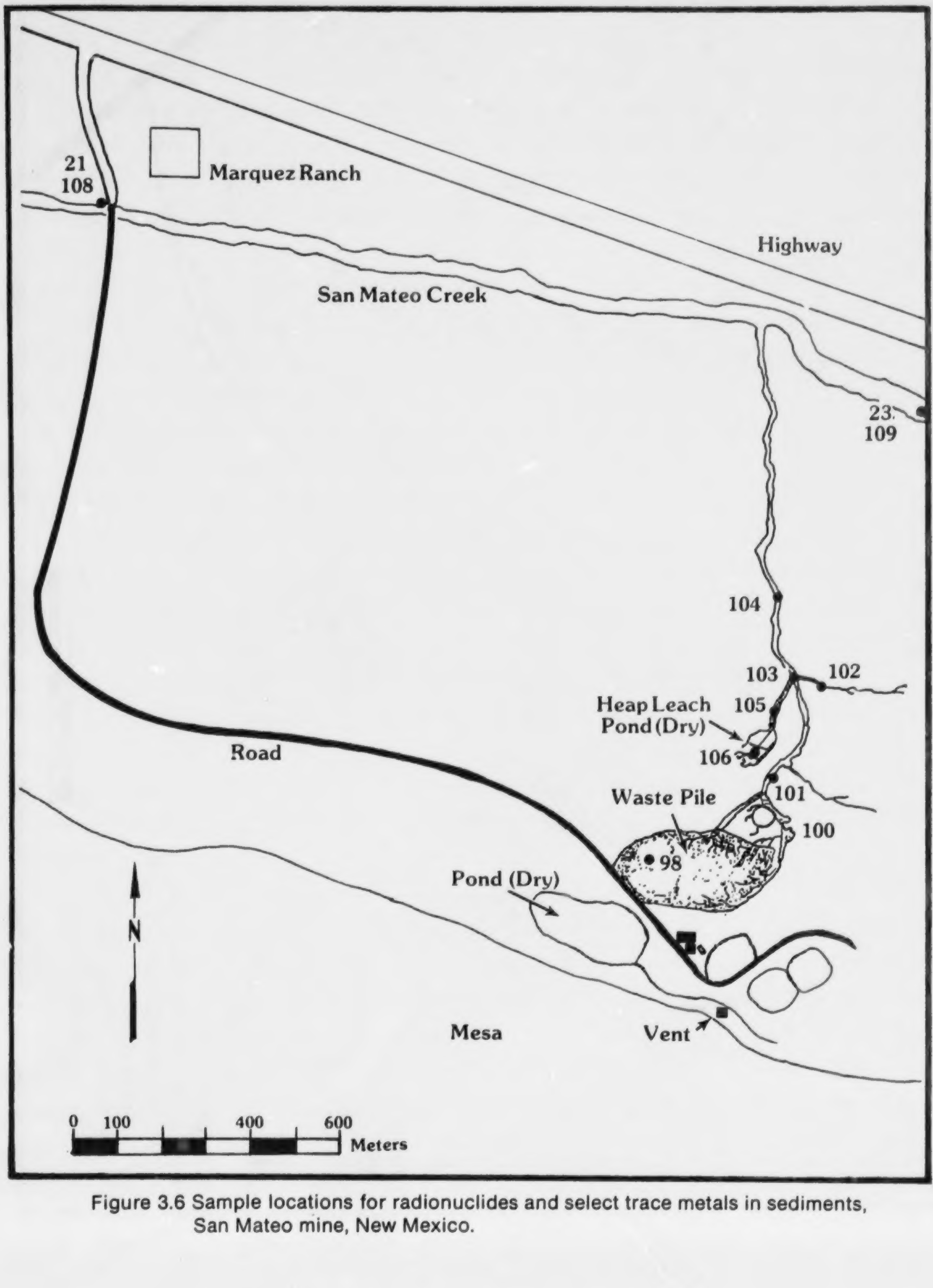


Figure 3.6 Sample locations for radionuclides and select trace metals in sediments, San Mateo mine, New Mexico.

ground site was detected. The Ra-226 content of the soil was about two times the background level at the furthest downstream site. This sample was collected about 130 m from the apparent source.

In summary, at the New Mexico inactive mine drainages the most prominent indicator of runoff from above-ground mineral storage is radium-226 in stream bed sediments. Concentrations in the source material are almost two orders of magnitude higher than those measured in the background soils. Elements such as uranium and selenium also have as large a concentration gradient, with concentrations decreasing downstream. At Poison Canyon, the radium-226 concentration diminished to two times background in a distance of approximately 100 m, while at the San Mateo site the distance was about 400 to 500 m. This may reflect either a more rapid transport by faster flowing water at the San Mateo site or, more likely, the larger source term there relative to background. At the San Mateo mine, radium-226 concentrations in water and sediments are significantly elevated downstream relative to upstream of the mine drainage.

3.2.3.3 Summary

The field studies conducted to date on the impacts of uranium mining on water quality are somewhat contradictory. Although no cases of gross, widespread contamination of groundwater or streams can be documented for uranium mining, there are cases of local contamination of water and sediments. From standpoints of theory and field data, there is need for cautious optimism in the use of local soil and water resources as sinks for waste discharge. Although numerous studies indicate that considerable reliance can be placed on the various physical and chemical processes to protect natural waters from contamination, investigations generally warn against using such studies to predict what may happen in other situations (Ru76; NRC79b; Ku79; Fu77; Am78). Laboratory results are highly dependent on the chemical properties of the fluid matrix and the physical and chemical properties of the particular soil studied. Results of field studies are site and time specific and have often suffered from inconsistent and undefined sampling and sample preservation techniques and questionable analytical measurements (Ku79; Ha78; Si77).

Our analyses reveal that there have been local water quality problems from mine water and wastes. Although widespread hazards have not been identified, this may be false security insofar as the present status of knowledge concerning trace element mobility in aqueous settings representative of uranium mining areas is rather unclear from both theoretical and real-data standpoints. Most often, effects of mining are interspersed with and masked by impacts from uranium milling. This complicates or renders impossible any meaningful interpretation of the mining-related data. Despite the attempt to sort out some of the information on trace element mobility, there is insufficient understanding at this time to dismiss or otherwise reduce the significance of trace element contributions (from mining activities) to surface streams and, to a lesser extent, to groundwater.

We conclude that there is considerable information on the topics of trace element chemistry. It is also clear that trace element concentrations in natural fresh water are highly variable on both macro and micro geographic scales. There is great difficulty in correlating concentrations with such characteristics as streamflow or lithologic environment. Accurate prediction of the behavior and cycling of trace elements through water and sediments first involves characterization of physical states such as particle size and form (chelate, colloid, complex ion, precipitate, etc.), speciation, and availability to plants and animals (An73). Andelman concludes "...that there can be large differences in trace element concentrations [in water], on both a macro and micro geographic scale, and that such variations often occur in an unsystematic and nonpredictable fashion."

We recommend additional studies of spatial and temporal variations, sources and sinks of trace elements, chemical interactions within the hydrogeologic system, interactions between surface and groundwater systems, effects on aquatic biota, and effects on water use (human consumption, stock watering, irrigation). Periodic monitoring in certain areas would allow for the detection of the long-term trends of potential changes that would accompany anticipated increases in future mining activity -- during a period of increased competition for scarce water resources.

3.3 Surface Mining

3.3.1 Solid Wastes

Surface mining consists of removing materials, separating them into ore, sub-ore, and overburden, and storing them in separate piles on the surface near the mine for various periods of time (Section 1.3.2). The various storage piles are managed differently, vary in size and level of contaminants, and exist for varying periods of time. All are potential sources of contamination to the environment via dusts suspended and transported by the wind, precipitation runoff, and Rn-222 emanation (Fig. 3.7).

3.3.1.1 Overburden Piles

Surface mining produces spoils at a rate of millions to tens-of-millions of tons per year. Unless this material is used to backfill the pit, large surface areas -- 40 hectares to over 400 hectares -- are covered to depths varying from a few meters to over 100 meters (Ka75, NRC77a, NRC77b, DOA78, Pe79).

Most of the mines begun since the early to mid 1970's use overburden to backfill mined-out areas of the pit (Ka75). Since older mines usually did not, erosion of their storage piles by water and wind may present an environmental problem (Ka75). In addition, the large amounts of overburden that past and present mines have used for road and dike construction and backfill also may present an environmental problem.

The annual average ore production of the 63 surface mines operating in the United States in 1978 was 1.2×10^5 MT (Section 1.3.1). Assuming an overburden to ore ratio of 50:1 (Section 1.3.2), the average annual production of overburden was about 6.0×10^6 MT per mine. A recent study of the eight large mines that accounted for 68 percent of the total 1977 United States U_3O_8 production from surface mines recommends the following average production parameters (Ni79):

1. ore production = 5.1×10^5 MT/yr
2. average ore grade = 0.11 percent U_3O_8
3. overburden:ore ratio = 77
4. overburden production = 4.0×10^7 MT/yr
5. mining days/yr = 330 d/yr

Surface areas of hypothetical overburden piles were computed using the above 63-mine and 8-mine overburden production rates and the following assumptions:

1. an average density of 2.0 MT/m^3 - reported values vary between 1.6 and 2.7 (Ro78, DOA78, NRC78a, Ni79)
2. the dumps are on level terrain
3. a rectangular waste dump with the length twice the width, and sides that slope at 45° angles (Fig. 3.8a)

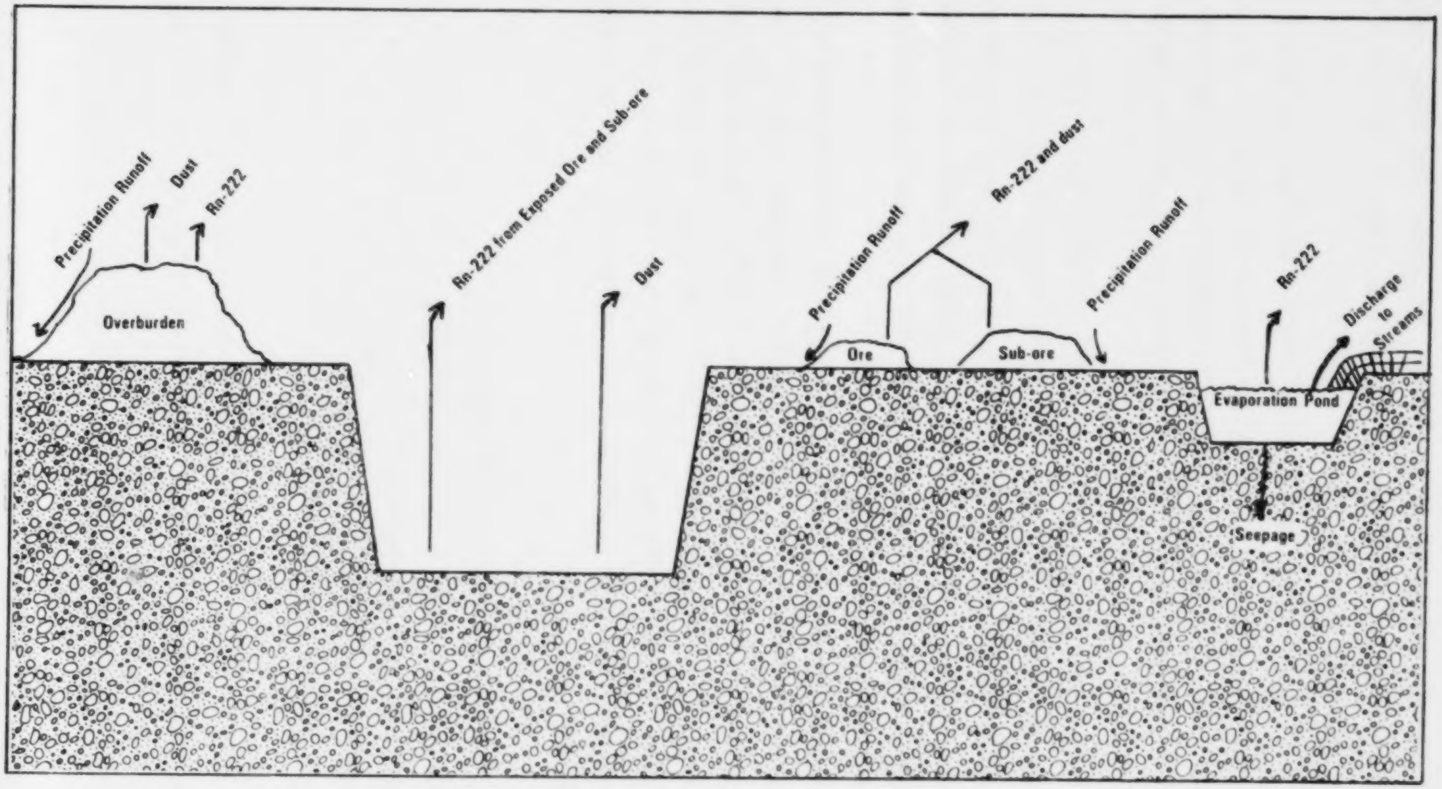


Figure 3.7 Potential sources of environmental contamination from active open pit uranium mines.

4. a waste dump in the shape of a truncated right-circular cone with 45° angled sides (Fig. 3.8b)
5. a bulking factor of 25% or 1.25 (Burris, E., Navajo Engineering and Construction Authority, Shiprock, NM, 2/80 personal communication)

Table 3.11 lists the surface areas of the hypothetical overburden piles in the following three cases:

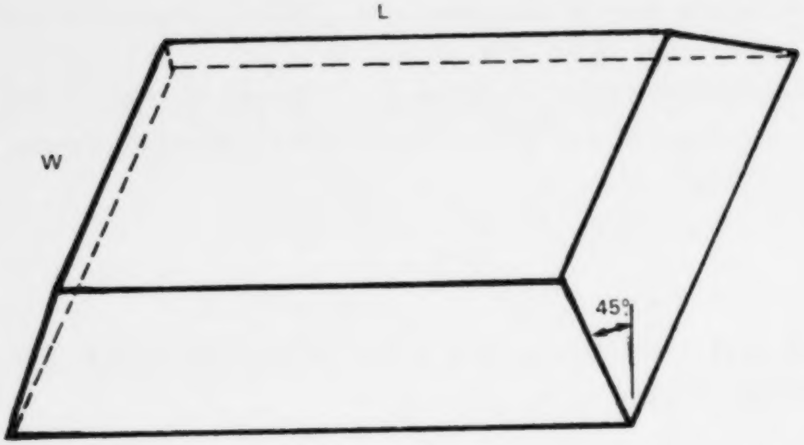
Case 1 - one year production with no backfilling

Case 2 - backfilling concurrent with mining - assumes 7 pits opened in a 17-year mine life with overburden from each successively mined pit used to backfill a previously completed pit, resulting in an equivalent of one pit of overburden (2.4-yr production) stored on the surface (Ni79)

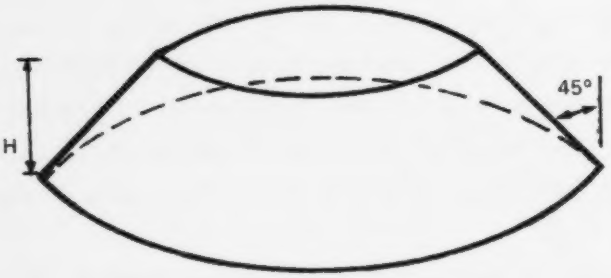
Case 3 - no backfilling during the 17-year mine life

The quantities of dust and Rn-222 that become airborne are directly proportional to the surface areas of waste piles. Table 3.11 shows the large variations possible between surface areas of waste piles at some active mines. Waste piles also cover various areas of terrain. However, for the same volumes, there are no significant differences in surface area or area of terrain covered for the two configurations of waste piles used in this study. Case 2 approximates recently activated mines, and Case 3 approximates older mines.

The type of rock in overburden spoil piles depends on the locations of the ore zones. Common rock types of New Mexico, Wyoming, and Texas mines include sandstone, claystone, siltstone, shale, and limestone, and unconsolidated silt, gravel, and sand (Co78, Pe79, Wy77, Ri78). In Texas, there are also lignite beds, tuffaceous silts, and some nearly pure volcanic ash



a) A rectangular pile with length twice the width and 45 degree sloping sides



b) A frustum of a regular cone with 45 degree sloping sides

Figure 3.8 Storage pile configurations assumed at surface and underground mines.

(Ka75). Coal veins are often present in Wyoming and New Mexico (Wy77, Ri78). However, the most abundant material in waste rock dumps will probably be clastic sedimentary rocks: sandstone, siltstone, and shale.

There is great variation in the particle size of material in waste piles, and this variation is important. Large particles (>30μm)*, because they usually settle within a few hundred feet of their origin, do not contribute to the airborne dust concentration (EPA77b). The potential for human respiration of the wind suspended dusts is also strongly influenced by the mean particle diameter (ICRP66).

Overburden rock is as large as available equipment can load and haul to the storage area. Rocks too large to handle with available equipment are broken into manageable sizes by small, explosive charges. Hence, rock particles will vary from less than a μm to a meter or more in diameter. Since weathering eventually breaks down the larger stones, the fraction of smaller particles increases over time.

Particle size distributions of material in waste rock piles at uranium mines have not been determined. It is likely that this material has a greater fraction of larger particles than that associated with crushed uranium mill tailings. Table 3.12 shows an example of the particle size distribution in the latter and the mean particle size distribution from a study of shale overburden removed from a surface mine in Pennsylvania (Ro78). Although the distribution fractions differ, a gross comparison can be made between the particle size of mill tailings and overburden waste. About 28 percent of the tailings were less than 50 μm in diameter, and only about 12 percent of the particles in the overburden pile had similarly small diameters. Because only particles smaller than 30 μm are likely to remain suspended by the wind for any significant distance (EPA77b), probably less than 10 percent of the overburden is a potential source of environmental contamination via wind erosion.

Table 3.13 shows the natural radionuclide concentrations in common rock types in the United States. In sedimentary rocks, which are common in the major uranium mining regions, the U-238 concentrations vary from less than 1 ppm** to about 4 ppm. Natural radioactivity usually is somewhat higher in the western states, and the uranium content in overburden prior to mining

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*μm = micrometer = 10^{-6} meters.

**ppm = parts-per-million = 10^{-6} grams per gram of rock.

Table 3.11 Estimated surface areas associated with overburden piles

Management[a]	Pile Height, m	Overburden Volume[b], m^3	Surface Area of Pile, m^2	Terrain Covered, Hectares
Rectangular Pile[c]				
		Average Large Mine[d]		
Case 1	65	2.5×10^7	5.2×10^5	48
Case 2	65	6.0×10^7	1.1×10^6	106
Case 2	30	6.0×10^7	2.2×10^6	209
Case 3	65	4.2×10^8	7.1×10^6	682
		Average Mine[e]		
Case 1	65	3.8×10^6	1.0×10^5	10
Case 2	65	9.0×10^6	2.2×10^5	20
Case 2	30	9.0×10^6	3.6×10^5	34
Case 3	65	6.4×10^7	1.2×10^6	113
Truncated Cone[f]				
		Average Large Mine[d]		
Case 1	65	2.5×10^7	5.2×10^5	46
Case 2	65	6.0×10^7	1.1×10^6	104
Case 2	30	6.0×10^7	2.1×10^6	208
Case 3	65	4.2×10^8	7.1×10^6	683
		Average Mine[e]		
Case 1	65	3.8×10^6	1.1×10^5	9
Case 2	65	9.0×10^6	2.2×10^5	18
Case 2	30	9.0×10^6	3.5×10^5	33
Case 3	65	6.4×10^7	1.2×10^6	110

Table 3.11 (continued)

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[a] Management:

Case 1 - one year production with no backfilling

Case 2 - backfilling concurrent with mining - assumes 7 pits opened in a 17-year mine life and equivalent of one-pit overburden (2.4 year production) remains on surface

Case 3 - no backfilling during 17-year mine life

[b] Volume = production (MT/yr) x production years x bulking factor (1.25) ÷ by density (2.0 MT/m^3).

[c] Length of pile is twice the width and the sides slope at a 45° angle (Fig. 3.8a).

[d] Overburden production = 4.0×10^7 MT/yr.

[e] Average 1978 overburden production of all 63 surface mines, assuming an overburden:ore ratio of 50/1, 6.0×10^6 MT/yr per mine.

[f] A frustum of a regular cone with 45° sloping sides (Fig. 3.8b).

is about 4 ppm (Ni79). However, during mining, some low-grade ore mixes with the overburden and may increase the concentration of the pile to as high as 20 ppm U_3O_8 (Ni79). This is equivalent to 12.6 disintegrations per minute (dpm) per gram of overburden. The progeny of the uranium will contribute additional radioactivity. Although there are local disequilibria between U-238 and its principal daughters, Th-230 and Ra-226, in ore-bearing rock, secular equilibrium will be assumed (Wo79). Small quantities of Th-232 and progeny will provide additional radioactivity. There is no apparent relationship between the Th-232 and U-238 decay chains. Th-232 concentrations in ores and host rock range from less than a pCi/g to a few pCi/g regardless of the U-238 concentration (Wo79).

Table 3.12 Particle size distributions of mill tailings and mine overburden

Mill Tailings[a]			Overburden[b]	
Particle Size, μm	Weight Percent	Conc.[c] / Avg. Conc.	Particle Size, μm	Weight Percent
250	60.3	0.15	>2000	75
125-250	7.5	0.03		
53-125	4.2	0.03	50-2000	13
44-53	3.8	0.03		
20-44	7.8	0.75	2-50	8
7-20	7.2	1.5		
1.4-7	9.1	4.6	<2	4
< 1.4	0.0			

(a)Source: Sc79.

(b)Source: Ro78.

(c)The concentration of radionuclides in that fraction divided by the average concentration.

Table 3.13 Natural radionuclide concentrations in various common rock types

	U-238		Th-232		K-40	
Rock Type	ppm	pCi/g	ppm	pCi/g	ppm	pCi/g
Igneous						
Basic	0.9	0.3	2.7	0.3	1.2	8.4
Granite	4.7	1.6	20	2.2	5.0	35
Sedimentary						
Shale	3.7	1.2	12	1.3	3.2	22
Sandstone	0.45	0.15	1.7	0.2	1.1	7.7
Limestone	2.2	0.7	1.7	0.2	0.32	2.2

Source: Oa72.

Table 3.14 shows the results of an extended airborne particle sampling program near a surface mine in New Mexico (Ea79). Although the on-site source of the radioactivity measured on these filters is undetermined, ore and sub-ore piles, waste rock piles, and mining activity all probably contribute. The higher activities reflect a greater contribution from ore dusts. From these air measurements, the above assumed average uranium concentration in overburden, 12.6 dpm/g (≅ 6 pCi/g), appears reasonable. These data also indicate that the progeny of U-238 through Ra-226 are in near secular equilibrium. The Th-232 concentration is about 1 pCi/g and, as indicated above, independent of the uranium concentration. Considering all available data, the radioactive source terms for overburden piles will be as follows: (1) U-238 and progeny = 6 pCi/g (0.0020 percent U_3O_8; (2) activity ratio (dust:overburden) = 2.5 (Section 3.3.1.2); and (3) Th-232 and progeny = 1 pCi/g. Figures 3.1 and 3.2 show the uranium and thorium decay series.

Table 3.14 Annual average airborne radionuclide concentrations in the vicinity of an open pit uranium mine, pCi/g

Location	U-238	Th-230	Ra-226	Th-232	U-238/Th-232
Jackpile Housing	76	80	70	1.2	63
Paguate	13	12	13	1.3	10
Bibo	9	7	5	1.3	7
Mesita	3	2	3	0.7	4
Old Laguna(a)	5	2	3	0.4	13

(a) Background location.

Source: Ea79.

Little information is available on stable element concentrations in overburden rock. Table 3.15 summarizes the analyses of a few grab samples of soil and rock from a uranium mine in New Mexico and one in Wyoming (Wo79). Except for possibly Se, V, and As, there are no significant concentrations of stable elements attributable to uranium mining. Considering the typically high natural Se and V contents of many minerals common to these areas and the limited number of analyses, the inference of pollution is indefinite. A relationship between uranium and the stable element concentrations does not appear to exist. Thus, the stable element concentrations in overburden from the model surface mine will be the average concentrations of samples 6, 7, and 8 in Table 3.15. Table 3.16 lists the average concentrations.

3.3.1.2 Ore Stockpiles

Ore is often stockpiled at the mine as well as the mill. Although ore stockpiles are much smaller than the overburden waste piles, the concentrations of most radioactive contaminants are much greater in ore-bearing rock than in overburden. In addition, ore is stockpiled at the mine for shorter periods of time than waste rock. Ore stockpile residence times vary from mine to mine and range from a few days to a few months. The recent study of 8 large surface mines cited 41 days as an average ore stockpile residence time (Ni79). We will use this value to estimate the average area of ore stockpiles.

The average of the 63 operating surface mines produced 1.2 x 10^5 MT of ore during 1978. Assuming 330 working days per year and a 41-day ore stockpile residence time, a 1.5 x 10^4 MT ore stockpile would exist at the average mine. In comparison, the recent Battelle study reported that the average of eight large surface mines produced 1550 MT of ore per day, which would yield a 6.3 x 10^4 MT ore storage pile, assuming the same residence time (Ni79). The ore piles vary in height at different mines and different times. One study reports a maximum pile height of 9.2 m (30 ft) (Ni79), and at another site the maximum and equilibrium ore pile heights are estimated to be 6.7 m and 3.1 m, respectively (NRC78a). Using these parameters and a bulking factor of 1.25 (Burris, E., Navajo Engineering and Construction Authority, Shiprock, NM, 2/80, personal communciation), the pile surface and pad areas were computed for the two production rates and two pile heights, 9.2 m and

Table 3.15 Uranium and stable element concentrations measured in rock and soil samples from two uranium mines

Sample	Concentration, µg/g														
	As	Ba	Cu	Cr	Fe$^{(a)}$	Hg	K$^{(a)}$	Mn	Mo	Pb	Se	Sr	V	Zn	U
Wyoming															
1. Top Soil Piles	3.2	700	13	46	1.3	< 4	2.2	190	2.9	23	<1	89	60	37	6
2. Sub-ore	<1.8	6800	9	<36	1.2	10	2.3	140	< 2.2	22	2.1	128	<100	25	61
3. Ore	5.4	800	9	<27	1.1	< 7	2.3	180	< 2.9	16	28	94	200	25	370
New Mexico															
4. Background Soil	4.1	450	12	<23	0.9	<4	1.8	200	5.5	12	<1	72	<50	22	<5
5. Background Soil	2.3	440	9	<20	0.8	<4	1.6	190	4.9	13	<1	50	<50	19	<5
6. Waste Pile	7.8	540	11	<28	0.8	< 5	1.4	260	2.5	10	<1	99	<70	23	8
7. Waste Pile	14	280	21	<43	0.7	< 8	0.5	750	<2.8	31	3.1	178	180	23	189
8. Sub-ore + waste	4.1	45	22	< 51	0.3	< 6	0.1	446	<1.8	25	< 1.4	179	<55	13	57
9. Ore	6.0	64	27	< 48	0.4	< 6	0.2	673	<1.8	31	1.5	323	<55	14	---

Source: Wo79.

(a) Units are percent.

Table 3.16 Concentration of radionuclides (pCi/g) and stable elements ($\mu g/g$) in overburden rock from the model surface mines

Element	Concentration	Element	Concentration
Arsenic	9	Selenium	2
Barium	290	Strontium	150
Copper	18	Vanadium	100
Chromium	<51	Zinc	20
Iron(a)	0.6	U-238	6
Mercury	<8	Th-230	6
Potassium(a)	0.7	Ra-226	6
Manganese	485	Pb-210	6
Molybdenum	2.5	Po-210	6
Lead	22	Th-232	1

(a)Units are percent.

3.1 m, assuming the same geometric configurations as for the overburden piles (Fig. 3.8). Table 3.17 gives the results. The computed surface areas of an average ore stockpile vary with volume of ore stored and pile height, but they are relatively independent of the pile shape.

Uranium deposits exist in sedimentary, metamorphic, and igneous formations. Sedimentary formations, primarily sandstone, siltstone, mudstone, and limestone generally host stratiform ore deposits often accompanied by carbonaceous material. Vein-type deposits usually occur in fractures of igneous and metamorphic formations. In the Rocky Mountain mining regions, about 98 percent of the recovered U_3O_8 comes from sandstone and related-type rock (St78). Sedimentary formations, principally sandstone, have been the predominant host for uranium in South Texas (Ka75).

Table 3.17 Estimated average areas of ore pile surface and pad

Pile Configuration[a]	Pile Height, m	Surface Area of Pile, m^2	Ore Pad Area, m^2
		Average Large Mine[b]	
Rectangular	9.2	6,300	5,700
Truncated Cone	9.2	6,200	5,300
Rectangular	3.1	14,000	13,500
Truncated Cone	3.1	13,700	13,200
		Average Mine[c]	
Rectangular	9.2	1,860	1,820
Truncated Cone	9.2	2,000	1,580
Rectangular	3.1	3,660	3,420
Truncated Cone	3.1	3,590	3,340

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[a] See Figure 3.8.

[b] Volume of ore = 6.3×10^4 MT (41 day production) x 1.25 (bulking factor) $\div$ 2.0 MT/m^3 = 3.9×10^4 m^3.

[c] Volume of ore = 1.5×10^4 MT (41 day production) x 1.25 (bulking factor) $\div$ 2.0 MT/m^3 = $9.4 \times 10^3 m^3$.

The DOE does not expect the mineralogical characteristics of uranium ore to change appreciably in the future, since the known reserves are mainly in sandstone or a related host (DOE79). This fact is apparent from the data in Table 3.18, which gives the distribution of ore reserves in the United States by type of host rock. More than 97 percent of the uranium reserves are in sedimentary formations, primarily sandstone. Hence, it is reasonable to assume that ore stockpiles in the future will continue to consist mainly of a friable (easily crumbled) sandstone rock.

No data are presently available on the particle size distribution of material in ore stockpiles. Thus, the particle size distribution of ore will be assumed to be similar to that of overburden rock.

The average grade of ore mined in 1978 was about 0.14 percent U_3O_8, but this will decline in future years (DOE79). The average grades of ore associated with the $30 and $50 reserves are 0.10 percent and 0.07 percent U_3O_8, respectively (DOE79). Assuming the average grade of ore mined in the next decade to be about 0.10 percent U_3O_8, the average uranium concentration in ore stockpiles will be 285 pCi/g (632 dpm/g). Although secular equilibrium in the uranium decay chain may not totally exist in some cases due to leaching by groundwater with subsequent redeposition, it appears reasonable to assume that radioactive equilibrium exists in a general assessment.

As discussed earlier, ambient Th-232 concentrations in the vicinity of a uranium mine range between 1 to 2 pCi/g. However, a concentration of 0.01 percent thorium is typical for ore from some surface mines (Mi76). This concentration is equivalent to 11 pCi Th-232/g of ore.

Uranium occurs in many ores as a secondary deposition. In a reducing environment, the soluble uranyl ion converts to insoluble uranium oxide and deposits preferentially on the smaller particles. (The total surface area of a given mass of smaller particles is greater than for larger particles.) Therefore, dusts that consist primarily of small particles have a greater specific concentration than ore as a whole (Table 3.12). The common procedure for computing uranium concentration in dust is to multiply the average concentration in the ore by 2.5 (NRC77a, NRC78a).

Table 3.18 Distribution of ore reserves by the type of host

Host Type	MT of Ore (10^6)	MT of U_3O_8	Percent Total Tons, U_3O_8
Sedimentary[a]	1,143.2	810,000	97.1
Lignite Materials	2.2	3,000	0.4
Limestone	1.3	1,200	0.1
Igneous and Metamorphic	32.7	20,400	2.4
Totals	1,179.4	834,600	100.0

[a] Principally sandstone, but includes conglomerates, shale, mudstone, etc.

Note.--The reserves are \$50 or less per pound U_3O_8, effective January 1, 1979 (DOE79).

Table 3.19 Average stable element concentrations in sandstone ores of New Mexico

Metal	Concentration, µg/g[a]	Metal	Concentration, µg/g[a]
Arsenic	86 (10-890)	Manganese	960 (70-3,000)
Barium	920 (150-1500)	Molybdenum	115 (3-700)
Cadmium	ND[b]	Nickel	20 (7-70)
Cobalt	16 (3-150)	Lead	78 (3-300)
Copper	61 (15-300)	Ruthenium	ND
Chromium	20 (7-70)	Selenium	110 (1-625)
Iron	15,700 (3,000-70,000)	Strontium	130 (1.5-300)
Mercury	ND	Vanadium	1410 (70-7,000)
Potassium	25,000 (7,000-30,000)	Zinc	29 (10-70)
Magnesium	3,500 (700-15,000)		

[a] Range of concentrations given in parentheses.

[b] ND - not detected

Note.--Ore samples are Dakota and Morrison sandstone from 25 uranium mines (Hi69).

In accord with the above discussion, we assume the following estimated average radionuclide source terms for ore stockpiles: (1) U-238 and progeny = 285 pCi/g ore (0.10 percent U_3O_8); (2) Activity ratio (dust:ore) = 2.5; and (3) Th-232 and progeny = 10 pCi/g ore.

Stable elements -- molybdenum, selenium, arsenic, manganese, vanadium, copper, zinc, and lead -- often associated with uranium ore at elevated concentrations may cause deleterious environmental and health effects. Mercury and cadmium are present only on rare occasions (Th78). However, as discussed above, there is no apparent relationship between concentration of stable elements and ore grade (Wo79). Table 3.19 lists measured (Hi69) concentrations of stable elements in 25 sandstone ores from New Mexico and average concentrations computed from these data. We assume the average concentration for the ore from the model surface mine.

3.3.1.3 Sub-ore Piles

All mines recover some rock containing uranium ore that at the time of mining is uneconomic to mill. The grade of this "sub-ore" varies with the "cutoff" level assigned by the mill. Some mines process sub-ore by heap leaching, which changes the chemical properties and constituents of the pile (Section 1.3.5.1). However, most mines store the sub-ore in separate piles and recover it when it becomes economically feasible.

The sizes of sub-ore dump piles vary with the quantity of ore mined and its grade. One study suggests that the sub-ore accumulation rate equals the ore production rate (Ni79), a ratio similar to that reported for the Sweetwater uranium mining operation (NRC77a). Using this assumption with the ore production rates given above for the average large mine and average mine, 5.1×10^5 MT/yr and 1.2×10^5 MT/yr, respectively, the average sizes of sub-ore piles generated at a constant rate during the 17-year active life of a mine were based on an 8.5 year accumulation and a bulking factor of 1.25. Figure 3.8 shows the shapes of the piles assumed, and Table 3.20 gives the results for piles 30 m high. The surface areas of the two pile configurations differ very little.

The mineralogical characteristics of ore and sub-ore are very similar. Thus, the distribution in Table 3.18 will apply to sub-ore. This study considers the particle size distribution of sub-ore the same as for overburden and ore.

In the early mining years, the ore cutoff grade was usually about 0.15 percent U_3O_8. However, this has continually decreased until today the cutoff ore grade is about 0.03 percent U_3O_8 (Ni79, NRC77a). Hence the ore content of these piles will be less than 0.03 percent U_3O_8, and the average content has been estimated to be one-half the cutoff grade, or 0.015 percent U_3O_8 (Ni79), which is equivalent to 43 pCi U-238/g (95 dpm/g). Also, the uranium in the sub-ore, as in ore, is assumed to be in secular equilibrium with its progeny. Because the occurrence of uranium in sub-ore is the same as in ore and the mineralogies are similar, the uranium in sub-ore should be concentrated on small particles by the same factor as in ore, 2.5.

The Th-232 concentration in sub-ore is between the ambient level and that in the associated ore, 1 pCi/g to 11 pCi/g. For lack of measured Th-232 concentrations, we assume that less than 2 pCi/g of Th-232 will be present (Table 3.14). The radiological significance of an error in this assumption will be small.

From the above discussion, we assume the following estimated average radionuclide source terms for sub-ore piles: U-238 and progeny = 40 pCi/g (0.015 percent U_3O_8); activity ratio (dust:sub-ore) = 2.5; and Th-232 and progeny = 2 pCi/g. Figures 3.1 and 3.2 show the uranium and thorium progeny.

Table 3.20 Estimated average surface areas of sub-ore piles during the 17-year active mining period

Pile Configuration(a)	Surface Area of Pile, m^2	Terrain Covered, Hectares
	Average Large Mine(b)	
Rectangular	1.2×10^5	11
Truncated Cone	1.2×10^5	11
	Average Mine(c)	
Rectangular	3.5×10^4	3.2
Truncated Cone	3.6×10^4	3.0

(a)See Fig. 3.8.

(b)Volume of sub-ore = 8.5 yr x 5.1×10^5 MT/yr x 1.25 ÷ 2.0 MT/m^3 = 2.7×10^6 m^3.

(c)Volume of sub-ore = 8.5 yr x 1.2×10^5 MT/yr x 1.25 ÷ 2.0 MT/m^3 = $6.4 \times 10^5 m^3$.

Stable elements observed in ore will also be present in sub-ore. Because stable element concentrations specific to sub-ore are unavailable and are unrelated to ore grade, concentrations in the sub-ore from the model surface mine will be assumed equal to those in the ore (Table 3.19).

3.3.1.4 Reclamation of Overburden Piles

Reclamation is usually done only for overburden piles. Ore stockpiles are continually being disturbed and their residence time is short. Also, sub-ore piles generally are not stabilized in anticipation of recovering the uranium at a later time. Hence, only overburden and waste rock piles are considered for stabilization and reclamation. Section 1.3.2 gives a brief description of these practices.

Backfilling mined out areas of the pit is necessary for an adequate reclamation program. Because of the swelling of earthern material once mined, sufficient material should be available to completely fill the pit when mining is completed. However, even though backfilling is generally being performed at most recently active mine sites, sufficient overburden is often not replaced to eliminate the pit.

Improperly stabilized spoil piles may become sources of contaminants to the environment. The wind can suspend and transport small-sized particles containing elevated levels of contaminants. Radon-222, produced by the radioactive decay of Ra-226 contained in the rocks, can emanate from the pile surfaces. Precipitation runoff from the piles can carry particulate matter and dissolved contaminants into the natural surface drainage system if rainfall exceeds the infiltration and holding capacity of the pile. The general procedure for reducing wind and water erosion is to grade the piles to conform to the natural terrain, cover the area with a layer of topsoil, and seed it with a native grass.

These spoils consist of unweathered and unconsolidated rock, coarse gravels, and sands and allied materials isolated from the natural processes that occur on surface soils. Consequently, spoils have poor textural properties and low water-holding capacities. Having no established flora to aerate the surface and make nutrients available, spoils are barren of nutrients required for plant growth. Hence, to sustain vegetation on these piles may be difficult because of poor soil quality and the arid conditions in the principal mining regions. Therefore, all plant growth depends on the topsoil cover, which is generally less than 30 cm thick (Re76). This is often inadequate to store sufficient water and nutrients to sustain plant growth during extended dry periods. Soil irrigation and fertilization may be required for several years until plants can sustain themselves.

Proper grading of the spoil piles, with water management and conservation, can help reclamation. The piles should have less than a 3:1 slope to reduce surface water runoff and erosion (St78). Forming catchment basins and terraces to hold water on the spoils and reduce water erosion will also increase the amount of runoff available to the plants. It also has been determined that vegetation on north-facing slopes requires about half the applied water of that on south-facing slopes (Re76). Water requirements of vegetation on horizontal surfaces and east and west slopes are about intermediate between those of the north and south slopes. Hence, spoil piles with long, north slopes will conserve water and reduce the irrigation required. Locating piles on leeward slopes and away from natural drainage will also reduce wind and water erosion.

The reestablishment of native grasses and shrubs is essential for controlling wind and water erosion and providing wildlife habitat. Wyoming requires a pre-mining vegetation inventory for use in evaluating post-mining reclamation (Wy76). Similar statutes governing mine reclamation are in effect in other states (Section 1.4). The Soil Conservation Service has recommended seed mixtures that are best suited to climatic and soil conditions in different areas of the West (St78). Newly seeded areas are usually protected from grazing by fencing for at least two growing seasons to allow the plants to become established.

Abandoned pits fill with water and form small lakes that livestock and wildlife can use for drinking water, if the water is uncontaminated. But, unless properly managed, final pits may be hazards to people and wildlife. Therefore, steep walls should be graded to give safe access into the pit, and after grading, the pit banks should be seeded to minimize erosion and prevent the sides from sloughing off.

3.3.2 Mine Water Discharge

3.3.2.1 Data Sources

The principal sources of information used to model the mining region in

Wyoming are the site-specific EIS's and ER's for active and proposed mining/milling operations and the NPDES permit data on discharge volume and quality. Several reports by state and federal agencies supplemented the foregoing, particularly with respect to estimating ambient water quality and flood volumes for various return periods and annual or monthly flows in principal streams of the region. Foremost among these is work by the State (Ha78), the U.S. Geological Survey (Cr78, Ho73), and the Soil Conservation Service (DOA75).

Self-monitoring data collected by industry and reported to EPA were also checked to ascertain compliance with NPDES permit conditions. Unfortunately, the permits do not specify limits on the volume of discharge; hence, the total mass or flux per unit of time may or may not agree with the values originally estimated by the discharges in the EIS, ER, or license application.

3.3.2.2 Quantity and Quality of Discharge

The purpose of this section is to identify water quality associated with surface uranium mining in the Wyoming Basin. This area was selected for detailed source term characterization and pathways analysis because of past and ongoing uranium production, primarily by surface mining. A subsequent section (3.4.2) similarly addresses underground mining. The analysis to follow is incomplete and preliminary, owing to the limited existing data, the lack of opportunity for significant new investigations in the time of this study, and the decision to pursue the objectives on a "model area/model mine" approach. So many variables of ore occurrence, mining practices, climate, geology, and hydrology exist that a detailed investigation is unrealistic.

Table 3.21 summarizes water quality data for seven surface and three underground mines in Wyoming. Uranium averages 0.62 mg/ℓ and ranges from 0.02 to 1.3 mg/ℓ. Dissolved radium-226 is typically less than 4 pCi/ℓ, although one mine reportedly discharged 10.66 pCi/ℓ. Suspended solids average 24.9 mg/ℓ. There is considerable variation from one facility to another; the observed range is 2.7 to 87.2 mg/ℓ. Zinc is the only stable element consistently monitored, probably because the NPDES permit addresses it. Concentrations average 0.04 mg/ℓ and are well below the 0.5 mg/ℓ limit in the permits. Barium and arsenic are less frequently monitored but appear to be in the range of 0.05 mg/ℓ for barium to 0.005 mg/ℓ for arsenic. Both of these values are well below the discharge limits.

Table 3.21 Summary of average discharge and water quality data for uranium mines in Wyoming and a comparison with NPDES limits

Project	Mine Type	Discharge m^3/min	Radioactivity Total U mg/ℓ	 Ra-226 pCi/ℓ	TSS	Major and trace constituents, mg/ℓ SO_4	 Zn	 Fe	 Ba	 Cd	 As
1	U	0.85	0.95	3.92	87.2		0.08	1.25			
2	U	6.57	0.41	2.28	2.7	234	0.02	0.02	0.05		
3	U	0.70	0.02	7.41	8.8		0.01				
4	S	1.89	1.30	10.66	5.0		0.01				
5	S	3.60	0.63	3.94	11.1		0.14				
6	S	5.68	0.02	2.85	10		0.05				
7	S	3.52	0.98	0.67	19.4	875	0.05			0.004	0.005
8	S	1.21	0.14	3.03	17.3		0.02				
9	S	0.10	1.14	3.6	62.5		0.16				
10	S	4.55									
All Mine Types (1 through 10):											
Average:		2.87	0.62	4.26	24.9	555	0.06	0.64	0.05	0.004	0.005
Standard Deviation:		2.25	0.50	3.00	29.5	453	0.06	0.87	-	-	-
Underground Mines (1 through 3):											
Average		2.71	0.46	4.54	32.9	234	0.04	0.64	0.05		
Standard Deviation:		3.35	0.47	2.62	47.1	-	0.04	0.87	-		
Surface Mines (4 through 10):											
Average		2.94	0.70	4.1	20.88	875	0.071			0.004	0.005
Standard Deviation:		1.96	0.53	3.4	21.04	-	0.063			-	-
Summary of NPDES Permit Limits Daily Average/Daily Maximum			2/4	3/10[a] 10/30 Total Radium	20/30		0.5/10	-/2	-/1	0.05/0.1	0.5/1

[a] Total Ra-226 limit is not monitored.

Source: NPDES permits from Region VIII (R. Walline, written communciation), site-specific reports (EIS, ER), and self-monitoring data.

Mean values from six surface mining projects in Wyoming were the basis for estimating the effects of mine discharge on water quality. Values from mines in the South Powder River Basin model area compare very well with the Wyoming mines, thus supporting adoption of a model mine in the Basin. There were no strong differences in water quality between surface and underground mines. Table 3.21 shows that discharge is highly variable, ranging from 0.1 to 6.57 m^3/min, with an average of 2.87 m^3/min. In surface mining projects, the average is 2.94 m^3/min, with a standard deviation of 1.96, indicating considerable discharge variation among facilities. This study assumes an overall average flow of 3 m^3/min from each surface mine in the calculations of chemical loading of local and regional streams (see Section 3.3.3 and Appendix H).

Table 3.22 shows water quality and flow rates associated with open pit mines in other areas and in various stages of operation. Ongoing development of an open pit mine in Colorado involves 28 m^3/min discharge and is therefore well above the average. Radium, uranium, and suspended solids are relatively low. Producing open pit mines in New Mexico are usually dry or nearly so and are dewatered at rates of 0.6 m^3/min or less. The water is used for dust control. Radium concentrations can be very high (New Mexico Projects) due to long residence time of groundwater in the ore body and the concentrating effects of evaporation. Similary, groundwater associated with ore bodies in Texas and Wyoming may contain several hundred picocuries per liter.

Mine dewatering has the greatest potential for adverse environmental and public health impacts. Although contaminant concentrations in the effluent conform to NPDES requirements, there is long-term contaminant loading to the ambient environment. Contaminants concentrate on stream sediments because of sorption and evaporation and become available for transport by flood water. Regional or at least local dewatering of ore bodies may deplete high quality groundwater. Theoretically, dewatering may induce horizontal or vertical influx of poorer quality groundwater into productive or potentially productive aquifers, but the extent of this phenomenon is poorly documented. We strongly recommend further study because the work done to date is largely oriented toward determining engineering feasibility versus the overall environmental impact.

Table 3.22 Water quality associated with surface and underground mines in various stages of construction and operation

Project	Discharge m^3/min	Total U mg/ℓ	Dissolved Ra-226 pCi/ℓ	Pb-210 pCi/ℓ	Milligrams per liter				
					TSS	SO_4	As	Mo	Se
Colorado									
Open pit mine:									
Development stage	28	1.044	4.10		16.2				
New Mexico									
Producing open pit mine, seepage to pit	0.13	2.5	180	17	168	2151	0.005	0.018	0.019
Open pit mine, ponded inflow water	0.58	2.6	220	26	23	842	0.005	0.545	0.043
Texas									
Active open pit mine holding pond.			50 to 100			380	< 0.01	< 0.01	< 0.01

Overland flow is not dismissed herein as a significant pathway, although its impact is of lesser importance according to data from April 1979 field studies in New Mexico and Wyoming (see Section 3.2.3.2 and Appendix G). A recent U.S. Geological Survey study for the Bureau of Indian Affairs (Ku79) addresses projected effects of runoff over long time periods if wastes and sub-ore are not stabilized or covered. The study concludes, with essentially no real data, that stream flows are too small in the sub-basin to transport wastes. In the larger basins, such as the Rio Puerco, sediment loads are so great that addition of tailings and, presumably, mine wastes would be insignificant. It is our opinion that additional field study is needed. Overland flow in a long time period could move radionuclides in the wastes into the main stream channels. Since this source will be available for many years after mine closure, if wastes are not stabilized, it may become a major one.

Seepage of contaminated water from mine holding ponds, which are operated to reduce suspended solids concentrations in mine discharge water, is believed to be insignificant. Since the ponds have relatively small areas, their seepage losses are small compared to losses by infiltration of releases to the watercourses. In some mining areas, such as the Powder River Basin, shallow groundwater quality is naturally poor. Maximum attenuation of contaminants is expected in the shallow, poorly permeable bedrock strata of the Wasatch and Fort Union Formations.

3.3.3 Hydraulic and Water Quality Effects of Surface Mine Discharge

3.3.3.1 Runoff and Flooding in the Model Surface Mine Area

3.3.3.1.1 Study Approach

Precipitation and runoff estimation for the model surface mine scenario in Wyoming considers three hydrographic units: sub-basin, basin, and regional basin. Respective surface areas are 11.4, 5,400, and 13,650 square kilometers (km^2). The mine is located in the sub-basin. The sub-basin, the basin, and the regional basin are all drained by ephemeral streams. The latter is drained by a major regional river that has wide seasonal variations in flow and is dry or nearly so about 180 days each year. The sub-basin has similar flow variability. Figure 3.9 depicts the mine in relation to the sub-basin, basin, and regional basin.

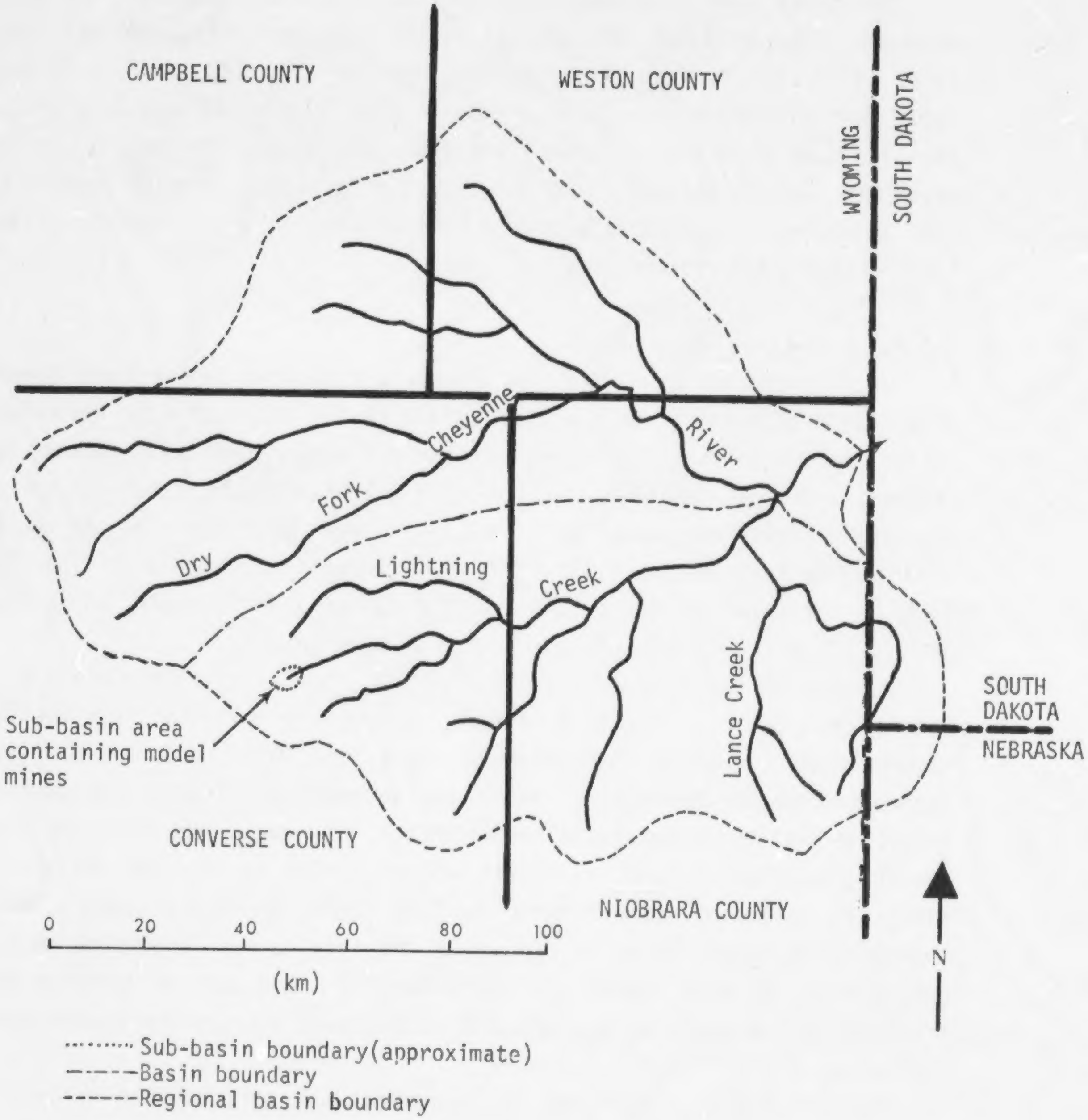

Figure 3.9 Sketch of sub-basin, basin, and regional basin showing orientation of principal drainage courses, areas of drainage, and location of mines.

The first general approach defined quality and volume of mine water discharge. Hypothetical hydrographic basins were then delineated and flood flows calculated for return periods ranging from 2 to 100 years. The individual and collective effects of discharge from three mines were then evaluated in terms of perennial flow, flood flow, and chemical transport. Of key importance was an estimation of the extent of perennial streams created by mine discharge and the influence of contaminants on water quality in the river draining the regional basin.

3.3.3.1.2 Description of Area

We selected an area of active mining and milling in the South Powder River Basin of Wyoming for analysis. The area has four active or imminently active uranium mills and a number of open pit mines. Available data on the geology, hydrology, and water quality of the area are sparse, but because of the mining and milling activity are relatively well known for a remote region like northeastern Wyoming. The study team chose one mining and milling project in the area for field investigation in April 1979; hence, additional data became available and are used herein as appropriate.

Terrain in the area has low rolling hills and an average elevation of 1414 m (MSL datum). Since the climate is not very different from that of nearby Casper, Wyoming, meteorological data from that station are fairly representative of the region. There are no relatively large seasonal and annual variations in precipitation intensity, frequency, and duration. Mean annual precipitation over a 30-year record period is 28.5 cm and occurs mainly as scattered thunderstorms in late spring and early summer. These thunderstorms supply 25 to 50 percent of the total annual precipitation and are usually of high intensity, short duration, and can be quite local. Potential pan evaporation averages 110 cm per year and greatly exceeds precipitation.

Streams in the study area are ephemeral and only exhibit measurable surface flow during snowmelt and heavy thunderstorm activity. Average total monthly flow for the period 1948 through 1970 for Lance Creek and the Cheyenne River at Spencer, Wyoming reveal distinct high- and low-flow periods in the year (Fig. 3.10). We believe that the streams represent the basin and regional basin hydrographic units used herein. Large watersheds usually exhibit measurable surface flow for about 180 days per year. Small watersheds, 30 to 40 square kilometers, may not flow at all for several consec-

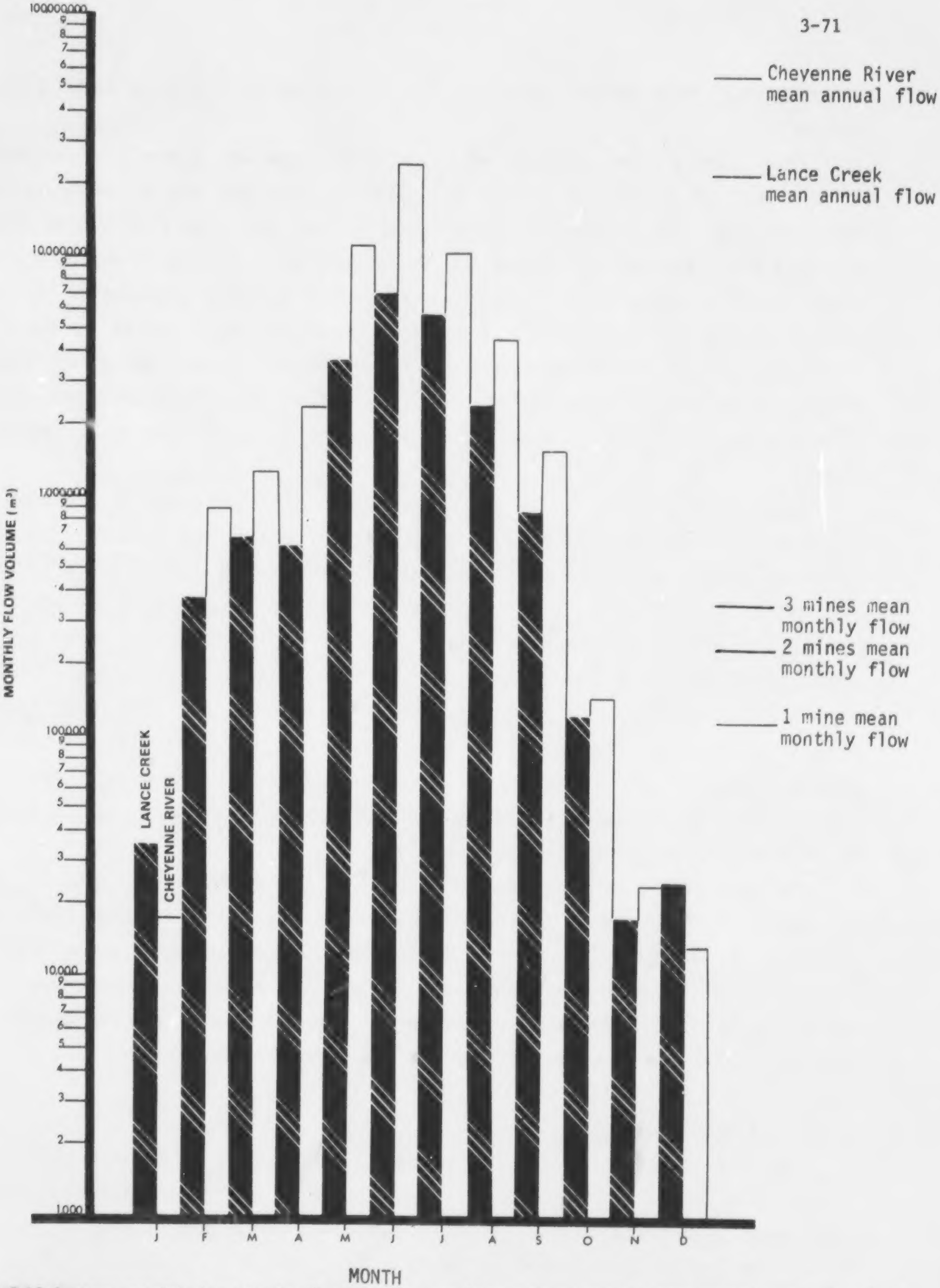

Figure 3.10 Average monthly flows for the Cheyenne River and Lance Creek near Spencer, Wyoming, for the period 1948-1970. (DOI59, DOI64, DOI69, DOI73)

utive years. Mean annual runoff is 0.8 to 1.3 cm or 0.0023 to 0.004 m^3/sec per km^2.

Peak flows in the regional basin and basin area are a result of snowmelt in at least 50 percent of the cases. This is commonly due to temporary but rapid melting from January to March. High flows can also result from widespread summer storms, but these are the exception. For small basins on the order of forty square kilometers or less, peak flows occur because of thunderstorms in the summer months. Thus peak flows in small basins versus the basin or regional basin commonly occur for different reasons and at different times in the year. A period of peak runoff from the sub-basin might coincide with a low flow or zero discharge condition in the basin or regional basin.

In the area of the Morton Ranch project (DOA75) there are 14 sub-basins, the average of which is about 11.4 km^2. Channel slopes are 11.4 to 31.4 m/km (average 21.2 m/km), and basin slopes are about 88 m/km. These are tributary to larger streams with channel slopes of 2.17 to 17.0 m/km (average 6.63 m/km), and which drain basins with an area of 5,400 km^2 and a mean annual flow of 0.80 m^3/sec. These, in turn, are tributary to a regional basin with an area of 13,650 km^2 and mean annual flow of 1.47 m^3/sec. All three hydrographic units are drained by ephemeral streams. The main stem of the regional system is dry an average of 180 days per year. The basin drains into the regional basin, assumed here to be the Cheyenne River Basin, which drains an area of 13,650 km^2 (Da75, Lo76, Ra77).

Surface water in the model area is used mainly for stock watering and irrigation. The amount of irrigated area in the basin is 1400 hectares, compared to 2800 in the regional basin. Because of extreme variability in surface flow volume and water quality, almost all municipal water comes from wells completed in bedrock. Stock water is from both wells and impoundments, whereas single-family domestic supplies are primarily from wells.

3.3.3.1.3 Method of Study

Because of dilution considerations, flow volume rather than peak discharge rate is of prime concern. For the basin and regional basin areas, only peak flow rate can be readily estimated on a probability basis for annual and longer time periods of perhaps 2, 5, 10, etc. years. Peak flow in the larger hydrographic areas commonly does not coincide with that in the smaller basins. Also, there is poor correlation between peak flow rate (Q) and total flow volume (V) for streams draining large basins. Total flow volume in the larger basins can be estimated from partial duration flow data. That is, we can estimate the percentage of the time, during the year, flow will be of a given magnitude.

Relationships among runoff volume, rainfall, and surface area in small basins (encompassing less than 30 square kilometers) in the Powder River Basin have been developed by the U.S. Geological Survey (Cr78) and the Soil Conservation Service (DOA75). Peak discharge and total annual flow in the basin and regional basin units were measured by the U.S. Geological Survey for Lance Creek at Spencer, Wyoming and for the Cheyenne River near Spencer.

We analyzed the effects of perennial or chronic mine discharge on changing existing ephemeral streams in the sub-basin, basin, and regional basin to perennial streams using a crude seepage and evaporation model. The basic equations and approach, explained in Appendix H, are similar to those used in the Generic Environmental Impact Statement on Uranium Milling (NRC79b). Adjustments were made for mine discharge rates and infiltration and evaporation losses. The main output of the model is an estimate of which stream segments might become perennial and what the net discharge would be from a number of mines operating in the same sub-basin. Water quality impacts can only be very roughly assessed. For the time being, we assume that infiltration and evaporation decrease flow but do not effect the chemical mass in the system. That is, we assume contaminants in mine drainage are deposited on or in the stream/wash substrate and remain available for transport by flood water.

The sub-basin is as shown in Fig. H.1 (Appendix H) and contains three active uranium mines, each of which discharges 4,320 m^3/day. Quaternary alluvium constituting the channel is assumed to have a porosity of 40 percent. The sub-basin contains seven streams or wash segments, three receiving mine water directly. Water from the mines dissipates by infiltration, evaporation, and as surface flow that may leave the sub-basin entirely. Appendix H shows the basic equations and assumptions and gives a complete summary of "losses" due to seepage and evaporation as well as any net outflow from the sub-basin.

Precipitation-runoff in the Wyoming study area correlates rather closely to basin size. Basins of about 10,000 km^2 area have an annual unit-area runoff of 0.43 cm/yr; whereas an area of perhaps 25 km^2 might have a runoff of only 5 cm/yr. Decreased runoff (on a unit area basis) associates with larger basins and reflects water storage, channel losses, and evapotranspiration that occur mainly in the tributaries. Impoundments are rarely on the main stem of streams, where washouts are a problem, but rather on tributaries. The average impoundment is located about every 130 square kilometers, is rather small, and is used for stock water. Very infrequently, small flood-irrigation projects may use impounded water for grasslands. Seventy-five percent of the annual runoff occurs during the summer thunderstorm activity in May, June, and July. Snowmelt occurs rather slowly and is captured in the headwater areas, whereas rainfall events are rather intense and localized, causing excess flows that reach the main stem, Lance Creek and Cheyenne River. Sediment loads are high in both the tributary and main stem streams.

Contaminant concentrations in overland and channel flow during peak runoff events in the sub-basin are expected to follow the pattern shown in Fig. 3.11, the data for which are from the U.S. Geological Survey (H. Lowham, in preparation) for a small basin, Salt Wells Creek, in the Green River Basin of southwestern Wyoming.

Note in the inset of Fig. 3.11 that the washoff peak, that portion of the runoff enriched in dissolved and suspended materials, precedes the runoff peak. Runoff in small basins is typically associated with brief but intense thunderstorms that flush the land surface. Total suspended solids (TSS) concentrations are disproportionately high in the peak flow events. Discharges of 170 m^3/min carry 100,000 mg/ℓ TSS; whereas flows of 1 cfs might carry only 500 mg/ℓ. The leading edge of the high flow has the greatest concentration of suspended solids and dissolved chemical load. Figure 3.12 depicts discharge and specific conductance values as a function of time for the same small basin in Wyoming. Specific conductance (SC) is a rough measure of the total dissolved solids (DS) content, following the approximate relationship: DS = 0.71 SC. Note that the first rise in the flow hydrograph occurs about three hours after the peak for specific conductance, indicating the presence of a contaminated "front" laden with salts and other suspended and soluble materials. The second peak on the flow hydrograph similarly precedes and is associated with degraded water quality due to this flushing

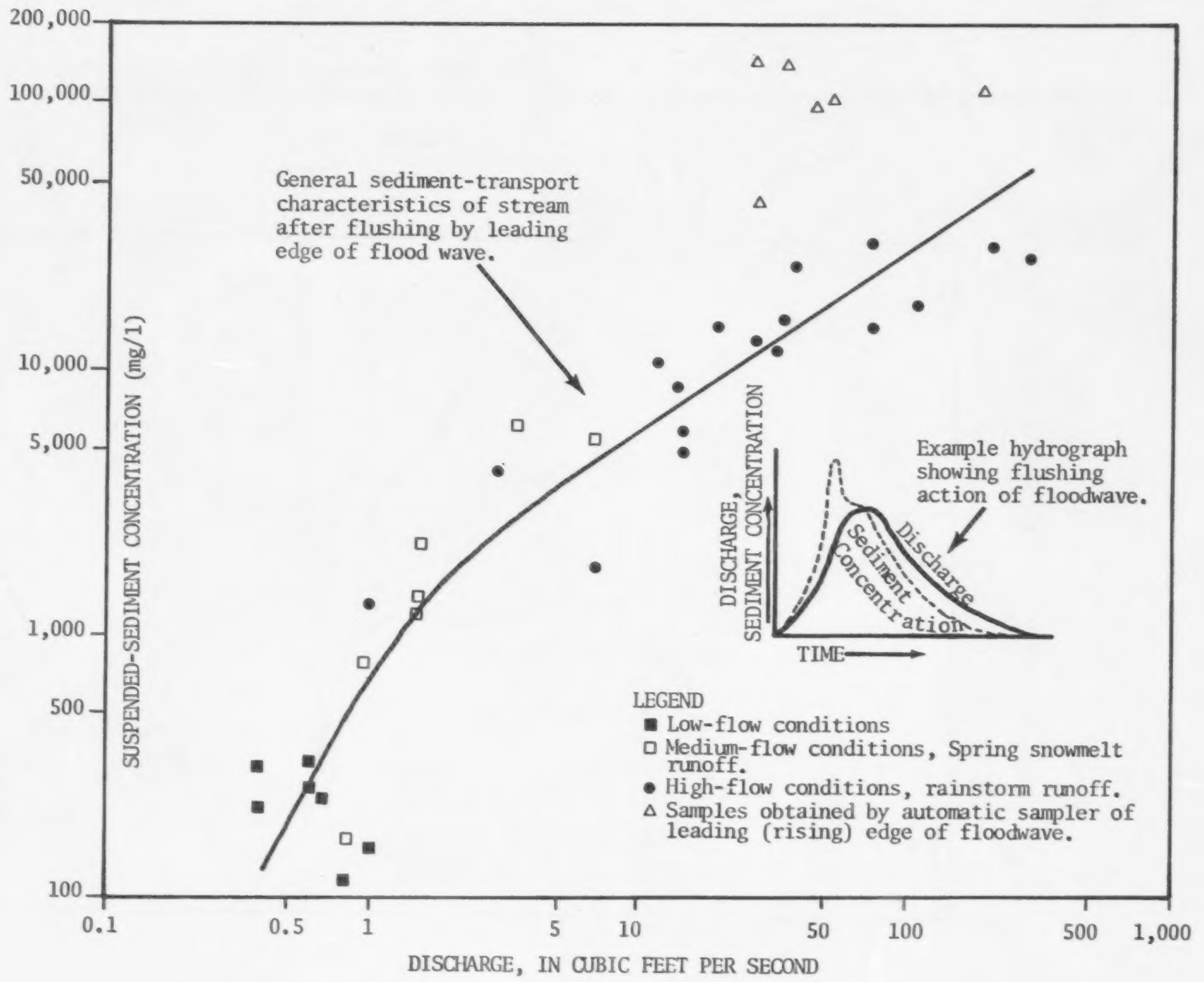

Figure 3.11 Suspended sediment concentration to discharge, Salt Wells Creek and tributaries, Wyoming (From U.S. Geological Survey data, H Lowham, in preparation)

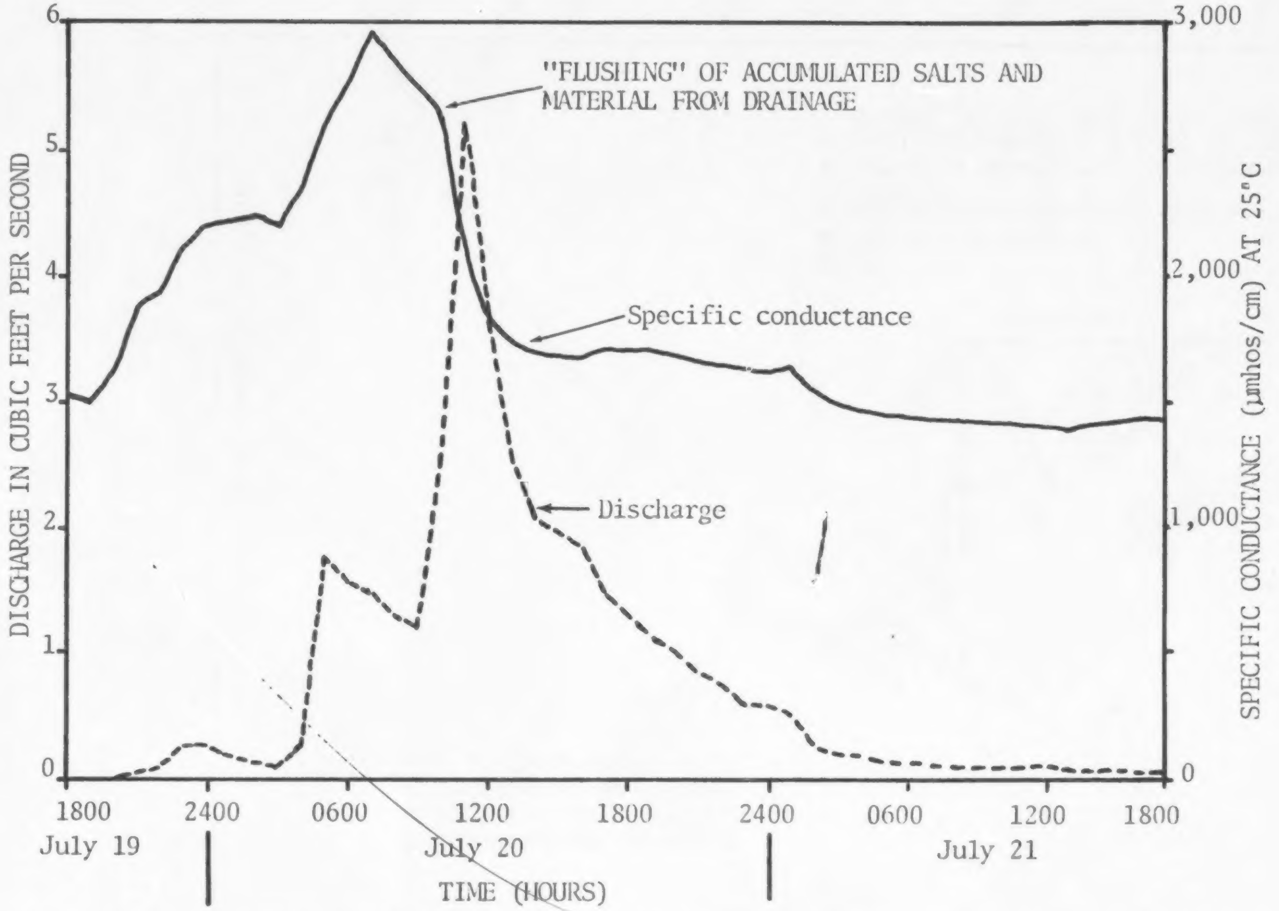

Figure 3.12 Relation of discharge and specific conductance to time at Salt Wells creek, Green River Basin, Wyoming. (From U.S. Geological Survey data, H. Lowham, in preparation.)

action. About 33 hours after precipitation begins, runoff water quality and flow very nearly approximate antecedent conditions. This indicates rather thorough flushing, most of which occurred in an 18-hour period.

Assuming similarity between the surface mining area and the situation described above, we believe intense flows of rather short duration flush most of the contaminants from the land surface and stream channels. Although sub-basin floods are expressed in terms of return period for Wyoming and in terms of partial duration (1-day, 7-days) and varying return periods for New Mexico, we believe the basic approaches (total flow vs. partial duration) to be rather similar because of the "flashy" nature of runoff in both study areas. In the New Mexico case, the mean discharge rate and the flow volume for the 7-day event are very often less than that for the 1-day event for the same return period. This also confirms the intense, short-term nature of runoff processes in the Wyoming and New Mexico model areas.

3.3.3.1.4 Discussion of Results

This section addresses the interaction between mine drainage and flood waters. Flood magnitude is addressed first, followed by calculation of water quality effects due to mine water. The U.S. Geological Survey technique (Cr78) for estimating floods in small basins in northeastern Wyoming was used to estimate peak discharge and total flow volume in the sub-basin. Multiple regression analysis reveals that the variables of area, slope, and relief provide roughly 90 percent correlation between rainfall and runoff (Cr78). Considering the numerous assumptions made throughout the analysis, only the area variable is used herein. It accounts for 70 percent of the flow. Table 3.23 shows the peak discharge rate and total flow volume from the sub-basin for floods with recurrence intervals (r) of 2 to 100 years. The basic equation for calculating discharge rate or flow volume is--

$$Q_r \text{ or } V_r = a\,A^{b_1} \qquad (3.1)$$

where a = regression constant

b_1 = drainage area coefficient for area: peak discharge area: volume relationships

A = basin area

= 11.4 km^2

Q_r, V_r = discharge rate and flow volume for flooding events with return periods of 2, 5, 10, 25, 50, 100 years.

The flow volume for the two-year flood is 32,921 m^3, and the instantaneous peak flow rate is 387 m^3/min. For comparison, we assume that the model mine discharges 3.00 m^3/min or 1.6 x 10^6 m^3 per year. Assuming the annual flood volume is 30,000 m^3, it is apparent that annual dilution is essentially nil and can be expected to be zero for perhaps 8 to 10 months of the year when there is no natural runoff. As Table 3.24 shows, flow volumes calculated using the Soil Conservation Service (DOA75) methodology average 1.5 times greater than those derived using the USGS approach. The latter are used herein in the interests of conservatism, i.e., there is less volume for dilution of contaminants.

Table 3.23 Peak discharge and total volume for floods of 2, 5, 10, 25, 50 and 100 year recurrence intervals

Recurrence Interval, r, in years	Regression Constant, a	b_1	Volume (m^3)	Regression Constant, a	b_1	Peak Discharge (m^3/min.)
2	9.62	0.689	32921	96.21	0.582	387
5	18.08	0.713	64116	199.6	0.612	840
10	24.87	0.727	90009	292.8	0.632	747
25	34.71	0.739	127862	441.1	0.660	1269
50	42.82	0.748	159920	575.4	0.679	2674
100	51.58	0.756	194937	731.1	0.699	3500

Peak flood discharges in the basin and regional basin range from 4,053 to 31,401 m^3/min for recurrence intervals of 25 years or less. For 50- and 100- year flooding events, peak discharge approximates 19,370 to 44,860 m^3/min. In the regional basin, discharge is 1.7 m^3/min or less approximately eight months of the year and equals or exceeds 17 m^3/min for about three months of the year, typically in the winter and early spring.

Maximum discharge from the basin and sub-basin is expected in the late spring and early summer months because of thunderstorms. At this time, flow in the river draining the regional basin is also at or near maximum, thus there is high probability for considerable dilution of runoff contaminated by mine drainage.

Total flow volumes for the basin and regional basin were estimated from U.S. Geological Survey records for the period 1948 to 1970. Figure 3.10 shows average monthly flows in cubic meters for the Cheyenne River and Lance Creek near Spencer, Wyoming. Immediately apparent is the close similarity in overall runoff pattern for the year.

Table 3.24 Summary of calculated total flow in the Wyoming model area sub-basin using the USGS and SCS methods

Recurrence Interval, r, in years	Sub-basin Total flow (m^3)(a)	Sub-basin Total flow (m^3)(b)
2	32,921	14,467
5	64,116	NC(c)
10	90,009	98,419
25	127,862	170,815
50	159,920	231,618
100	194,937	295,257

(a)Source: Cr78.
(b)Source: DOA75.
(c)NC = Not calculated.

Minimum flows occur in November, December, and January, and peak runoff in both basins occurs in May, June, and July. Long-term average annual flow in the basin is 2.18×10^7 m^3 and 5.64×10^7 m^3 in the Cheyenne River. These are almost exactly proportional to the respective basin areas of 5,360 km^2 and 13,650 km^2, indicating similar climatic and runoff conditions.

Assuming there are 3 mines operating for a 17-year period and that each mine discharges on the average 3.00 m^3/min continuously, total annual flow volume from the mines is 4.7 x 10^6 m^3. Cumulative discharge from the sub-basin is 7.04 m^3/min or 3.7 x 10^6 m^3/yr, which causes development of a perennial stream 12.8 km long within the basin. Insofar as the basin channel length is 141 km, the perennial stream ceases to flow well within the basin.

Appendix H explains the methodology and intermediate steps involved in deriving these foregoing values. Mine drainage water is not expected to flow the full length of Lance Creek or reach the Cheyenne River. However, on the basis of total monthly flow, the volume of mine drainage from one mine exceeds the flow in Lance Creek and the Cheyenne River for three months of the year, whereas flow from three mines exceeds basin flow for five months and regional basin flow for four months each year (Fig. 3.10).

The aqueous pathway for mine drainage is considered in terms of chronic, perennial transport in the mine water, per se, and transport by flood waters that periodically scour the channels where most of the sorbed contaminants would be located. Considering the random nature of flooding and the resulting uncertainty as to when the next 2-, 5-, or 10-year, etc. flood may occur, it is assumed that most contaminants accumulate on an annual basis and are redissolved by floods of varying return periods (2 to 10 years) and volumes. Many combinations of buildup and flooding are possible, such as buildup for 5 years or 10 years with perhaps several 2-year storms and one 5-year storm. Insofar as numerous assumptions are made in calculating volume and quality of mine discharge, basin runoff, and fate of the contaminants in the aqueous system, use of annual accretion and varying flood volumes in the sub-basin is considered adequate for estimating flood water quality.

Dilution of contaminated flows originating in the sub-basin and extending into the basin were conservatively calculated by assuming that the total flow during the low period equaled the mean annual flow. Thus, high flows and associated increased dilution are ignored, tending to make the analysis conservative. Contaminated flows from the sub-basin are diluted into these adjusted mean annual flows. Definition of the source term on an annual basis is most compatible with the radiation dose and health effects calculations in Section 6. Use of the low flow segment of the total annual flow regime is decidedly conservative since total flow during the five months of low flow conditions amounts to 111,610 m^3 and 218,336 m^3 for the basin and regional basin, respectively. Average annual flow for the period of record (22 years) is considerably higher, amounting to 2.184 x 10^7 m^3 for the basin and 5.64 x 10^7 m^3 for the regional basin.

Runoff in the basin and regional basin is expected to markedly dilute contaminated flood flows originating in the basin. Such floods would scour contaminants from about 23 kilometers of channel affected by contaminants from the three active mines. Peak runoff events in the sub-basin are most likely in the late spring-early summer season when runoff in the basin and regional basin is the maximum or near maximum, on the average. However, peak runoff from the sub-basin could also occur when the basin and regional basin are at low flow or zero discharge. Such contrasts are present between the basin and regional basin flow regimes. From September through December, Lance Creek can be expected to have no discharge from 45 to 65 percent of the time, whereas the Cheyenne River will be dry, on the average, from 65 to 85 percent of the time (Fig. 3.13). Thus there is a distinct chance that contaminants transported in Lance Creek would not be immediately diluted upon reaching the Cheyenne River.

Before discussing the calculated concentrations of contaminants in the basin and regional basin streams, several other conditions need to be mentioned. In water-short regions like Wyoming, extensive use is made of impoundments to capture and store runoff. On Lance Creek, the model for the basin, the volume of existing impoundments is 15.78 x 10^6 m^3 or 72 percent of the annual average runoff. In the regional basin, modeled after the Cheyenne River, there are 4.2 x 10^7 m^3 of storage volume, which is 74 percent of the average flow of 56.4 x 10^6 m^3. Thus, it is very likely that discharge from the sub-basin or basin will not exit the basin, particularly in the periods of low flow. Contaminant concentrations, particularly those affected by sorption and precipitation reactions, are likely to be reduced as a result of sedimentation and long residence time in the impoundments, although there is some potential for overtopping, disturbance by cattle, and so on. Significant adverse impacts are not likely considering precipitation and sorption reactions which are likely to remove contaminants from the food chain. Proof of this is lacking and we recommend confirmatory studies for the stable elements. Previous studies (Ha78; Wh76) emphasized radiological contaminants.

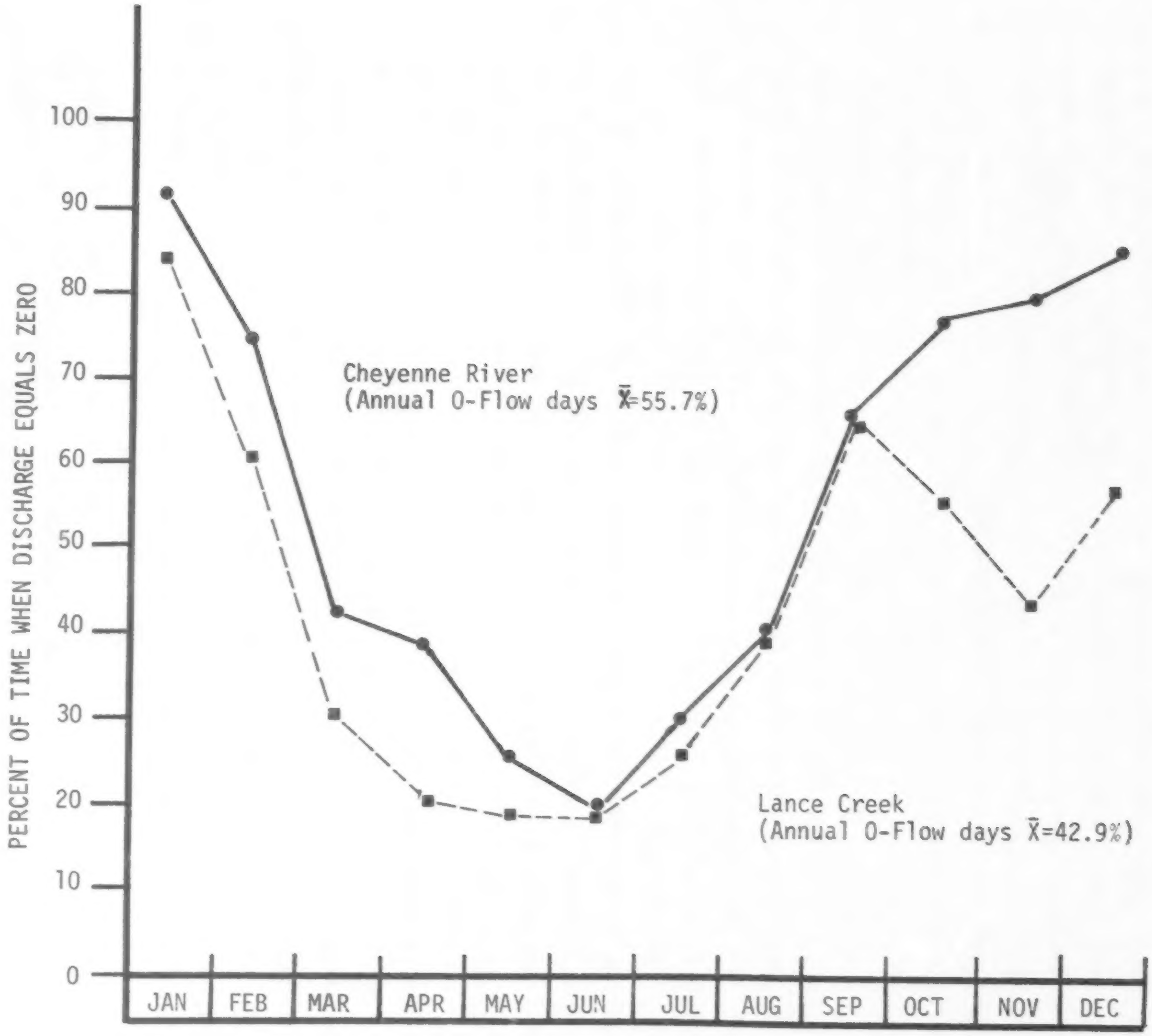

Figure 3.13 Periods of no flow in Lance Creek and the Cheyenne River near Riverton, Wyoming for the period 1948-1978 (Summarized from flow records provided by H. Lowham, U.S. Geological Survey, Cheyenne, WY.)

Radium-226 is strongly sorbed onto stream sediments and (or) it is subject to precipitation. Partial re-solution in subsequent floods occurs but it is assumed that only 10 percent of the mass deposited on an annual basis goes back into solution in flood waters. The rationale for this assumption is based on laboratory studies (Sh64; Ha68), field data from New Mexico (Ka75; Ku79), and review of the literature. Pertinent field and laboratory data specific to surface water quality in the Wyoming uranium mining areas are scarce, although studies by the State (summarized by Harp, 1978) are noteworthy. Sulfate is regarded herein as rather mobile and, as such, most of it infiltrates the shallow aquifer. Therefore, only 20 percent of the mass from a given mine on an annual basis is assumed available for re-solution in flood waters. The fate of zinc, arsenic, and cadmium is insufficiently understood to predict what fraction in the mine discharge will be removed from solution versus remain available for re-solution. Studies along these lines are necessary. Similarly, not all of the contaminants potentially present in mine waters from Wyoming are necessarily shown in Tables 3.21 and 3.25, which were developed based on available data from NPDES permits, environmental reports, and environmental impact statements. In the case of suspended solids, there is no calculation of non-point source contributions from mined lands. Sediment loads from such sources could be locally significant, but mined land reclamation and natural recovery seems to effectively mitigate problems. Only suspended solids from mine drainage, per se, are considered.

Table 3.25 shows the flood flow volumes (in the sub-basin) associated with events having return periods of 2, 5, 10, 25, 50, and 100 years. Also shown are the contaminant concentrations calculated from the annual contaminant loading diluted into the foregoing floods. As expected, concentrations are high because of the low dilution volumes associated with the small sub-basin. Surface water in the sub-basin might be impounded therein for use by stock or, less possibly, irrigation, but it is more likely that the principal impoundments would be in the larger hydrographic unit, the basin. The flood flow volumes shown represent runoff from the entire sub-basin. When the second and third mines begin to discharge, the annual loading and concentration values shown would have to be doubled or tripled. The reader should remember that background concentrations already present in flood runoff would be additive to the values in Table 3.25. However, these have been assumed

Table 3.25 Annual contaminant loading from one uranium mine and resulting concentrations in floods within the sub-basin for return periods of 2 to 100 years

Contaminant and concentration in mine effluent	Chemical mass available for transport on an annual basis	Flood flow volumes (m^3) and contaminant concentrations associated with return periods of 2 to 100 years[c]					
		V_2 = 32921 C_2	V_5 = 64116 C_5	V_{10} = 90009 C_{10}	V_{25} = 127862 C_{25}	V_{50} = 159920 C_{50}	V_{100} = 194937 C_{100}
Total uranium 0.070 mg/ℓ	110 kg/yr	3.34	1.72	1.22	0.86	0.69	0.56
Radium-226 4.1 pCi/ℓ	0.00065 Ci/yr[a]	19.7	10.1	7.2	5.1	4.1	3.3
Total suspended solids 20.9 mg/ℓ	32,955 kg/yr	1001	514	366	258	206	169
Sulfate 875 mg/ℓ	275,940 kg/yr[b]	8381	4304	3066	2158	1723	1416
Zinc 0.071 mg/ℓ	112.0 kg/yr	3.40	1.75	1.24	0.876	0.700	0.575
Cadmium 0.004 mg/ℓ	6.31 kg/yr	0.192	0.098	0.070	0.049	0.039	0.032
Arsenic < 0.005 mg/ℓ	7.88 kg/yr	0.239	0.123	0.088	0.062	0.049	0.040

[a] Ten percent of the annual loading is assumed available for solution. The balance is assumed sorbed onto sediments or present in insoluble precipitates.

[b] Twenty percent of the annual loading is assumed available for transport and the balance is assumed to have infiltrated to the water table or it is present as an insoluble precipitate.

[c] V_r and C_r refer to, respectively, flood volume, in cubic meters, and concentration in milligrams per liter or picocuries per liter for an r-year flood. Concentrations are in milligrams per liter except radium-226, in pCi/ℓ.

__Note__.--Assumptions: Mine discharges continuously at a rate of 3.00 m^3/min and concentrations are the average of those shown in Table 3.21. All suspended and dissolved contaminants remain in or on the stream sediments and are mobilized by flood flow.

equal to zero in order to estimate incremental increases due to mining and to simplify the calculations.

Table 3.26 shows contaminant concentrations in the basin and regional basin streams from the discharge of one mine. For cases involving two or more mines, the concentration shown would be scaled up by a factor of two or more. Basically, the table shows the effects of taking contaminated flood waters from the sub-basin and diluting them in the low flow volume of the basin and regional basin. As expected, concentrations decrease with floods of greater volume and longer return period. Additional dilution occurs when discharge from the basin enters the regional basin. Taking the two-year runoff event in the sub-basin, for example, uranium is diluted from 3.34 mg/ℓ (Table 3.25) to 0.76 mg/ℓ in the basin and then to 0.44 mg/ℓ in the regional basin. There is some question as to whether the lesser sub-basin floods, particularly those with return periods of 25 years or less, would actually flow the length of the basin and enter the regional basin. Because much of the 22.7 km reach of stream directly affected by mine discharge is located in the basin, it is conservatively assumed that the contaminants will reach the basin and eventually the regional basin. The foregoing analysis is structured as a worst-case, maximum-concentration scenario.

Concentrations of contaminants in flood waters affected by mine drainage are compared to water standards for potable and irrigation uses (Table 3.27). Radium-226 concentrations in the basin and regional basin streams (Table 3.27) range from 1.6 to 4.5 pCi/ℓ and are below the drinking water standard (for Ra-226 + Ra-228) of 5 pCi/ℓ. Uranium concentrations range from 0.26 to 0.76 mg/ℓ, which is roughly equivalent to 176 to 514 pCi/ℓ. On the basis of chemical toxicity alone, such concentrations would probably present no problem for short periods, but radioactivity is another matter. Reevaluation of the standard for uranium in potable water is presently receiving attention within the Agency (R. Sullivan and J. Giedt, USEPA, oral communication, 1980). Briefly, there is consensus that the radiotoxicity of uranium is similar to that of radium-226 and 228. For continuous ingestion at a rate of 2 liters per day, it is suggested that potable water contain no more than 10 pCi/ℓ (0.015 mg/ℓ) natural uranium to reduce the incidence of fatal cancers to no more than 0.7 to 3 per year per million population (Office of Drinking Water guidance to the State of Colorado, July 7, 1979). Realizing that the

Table 3.26 Concentrations in basin and regional basin streams as a result of surface mine discharge

Parameter	Concentrations (mg/ℓ ; pCi/ℓ in the case of radium) in basin discharge under low flow conditions due to influx of sub-basin floods with 2, 25, and 100 year return periods[(a)]			Concentrations (mg/ℓ ; pCi/ℓ in the case of radium) in regional basin discharge under low-flow conditions due to influx of basin discharge, also under low-flow conditions, and sub-basin floods with 2, 25, and 100 year return periods[(b)]		
	C_2	C_{25}	C_{100}	C_2	C_{25}	C_{100}
Total Uranium	0.76	0.46	0.36	0.44	0.32	0.26
Radium-226	4.5	2.7	2.1	2.6	1.9	1.6
Total Susp. Solids	228	138	107	131	95	79
Sulfate	1909	1152	900	1098	797	668
Zinc	0.774	0.468	0.366	0.445	0.324	0.271
Cadmium	0.044	0.026	0.020	0.025	0.018	0.015
Arsenic	0.054	0.033	0.025	0.031	0.023	0.019

(a) Calculated as follows: Assuming a two year flood, uranium concentration in the outflow from the sub-basin equals 3.34 mg/ℓ and flow equals 32,921 m^3 (see Table 3.25). Average total flow for 5 months of low flow conditions in the basin equals 111,610 m^3. The concentration in the basin outflow, after dilution of the contaminated inflow from the sub-basin for floods of varying recurrence intervals equals:

$$C_{Basin} = \frac{V_{Sub\text{-}basin} \times C_{Sub\text{-}basin}}{(V_{Sub\text{-}basin} + V_{Basin})} = \frac{(32921\ m^3)\ (3.34\ mg/\ell\ )}{32921\ m^3 + 111610\ m^3)} = 0.76\ mg/\ell$$

(b) Calculations similar to "a" above, except average total flow volume for 5 months of low flow in the regional basin equals 218,336 m^3. Hence,

$$C_{Regional\ basin} = \frac{V_{Sub\text{-}basin} \times C_{Sub\text{-}basin}}{(V_{Sub\text{-}basin} + V_{Regional\ Basin})}$$

Table 3.27 Comparison of potable and irrigation water standards and surface water quality affected by surface mine drainage

Parameter	Range of contaminant concentrations in flood flow affected by mine discharge[a]				Potable water standards[b]		Irrigation[c]
	Basin		Regional Basin		Maximum Permissable Concentration (mg/ℓ)	Recommended Limiting Concentration (mg/ℓ)	Recommendations for maximum concentration for continuous use on all soils (mg/ℓ)
	Min.	Max.	Min.	Max.			
Total U	0.36	0.76	0.26	0.44	0.015/3.5/0.21[d]	---	---
Ra-226 + 228	2.1	4.5	1.6	2.6	---	5 pCi/ℓ	5 pCi/ℓ
TSS	107	228	79	131	---	---	---
Sulfate	900	1909	668	1098	---	250	200
Zinc	0.366	0.774	0.271	0.445	---	5.0	2.0
Cadmium	0.02	0.044	0.015	0.025	0.01	---	0.010
Arsenic	0.025	0.054	0.019	0.031	0.05	0.01	0.10

[a] Concentrations in milligrams per liter, except Ra-226 -228 which are in picocuries per liter.

[b] Sources: U.S. Environmental Protection Agency (EPA76) and, in the case of uranium, suggested guidance from the National Academy of Sciences (NAS79) to the USEPA and from USEPA (Office of Drinking Water) to the State of Colorado (La79).

[c] Source: NAS72.

[d] 0.015 mg/ℓ : Suggested maximum daily limit based on radiotoxicity for potable water consumed at a rate of 2 liters per day on a continuous basis.

3.5 mg/ℓ : Suggested maximum 1-day limit based on chemical toxicity and intake of 2 liters in any one day.

0.21 mg/ℓ : Suggested maximum 7-day limit based on chemical toxicity and intake of 2 liters per day for 7 days.

limit of 10 pCi/ℓ (0.015 mg/ℓ) may not be cost effective, the Agency is contracting to develop the economic and technical basis for a uranium (in water) standard. The National Academy of Science, at the request of the Agency, evaluated the chemical toxicity of uranium. A maximum, 1-day concentration of 3.5 mg/ℓ (7 mg/day based on daily intake of 2 liters) is the "Suggested No Adverse Response Level" (SNARL). The corresponding concentration for a 7-day period is 0.21 mg/ℓ.

There are numerous complicating factors surrounding the foregoing suggested radiotoxicity and chemical toxicity limits for uranium. These include economic justification, technical feasibility, gut to blood transfer factors, and overall health of the receptor, to name a few. Of importance is the fact that a stricter standard for uranium in water is likely and that present NPDES limits of 1 mg/ℓ or previous drinking water limits on the order of 5 to 8 mg/ℓ are or will be superseded. For these reasons, the calculated uranium concentrations in the aqueous pathway are considered relative to the more recent, suggested limits of 0.015, 3.5, and 0.21 mg/ℓ.

Although mine effluents are not considered potable water, they infiltrate shallow aquifers that are potable in terms of the Safe Drinking Water Act. The extent to which shallow aquifers in uranium mining areas are used for potable water supply is presently small, but accurate surveys of well locations and water quality are scarce. If a limit of 0.015 mg/ℓ for uranium in potable water is set, it appears that uranium instead of radium-226 may be the primary pollutant of concern in both surface runoff and related shallow groundwater.

Of the remaining contaminants, sulfate and possibly cadmium might exceed drinking water standards. Cadmium may also limit use of the water for irrigation. These results provide only a rough estimate of water quality effects. There are other stable toxic elements to consider, but there are insufficient data. Multiple mine sources would increase the concentration, but ion exchange, sorption, etc. would reduce them. The net effect is simply unknown. It does appear that uranium, in particular, deserves additional study in light of new interpretations concerning radiotoxicity.

3.3.3.2 Impacts of Seepage on Groundwater

The previous analysis assumed no infiltration (to groundwater) of dissolved or suspended contaminants, thereby creating a maximum or worst-case situation with respect to transport via floodwaters. In fact, contaminants will also infiltrate through the stream deposits. Anions and selected stable elements like uranium, selenium, and molybdenum are most likely to migrate downward. Insofar as the alluvial, valley fill aquifer may be used locally, particularly in the case of larger drainage basins and the regional basin, some analysis of potential impacts is offered herein.

Effects of mine drainage impoundments used to settle suspended solids are excluded from the present analysis. Such impoundments are relatively small, commonly less than 1 or 2 hectares, and tend to become self-sealing due to settling of fines. Potable water supplies at the mines are usually from deep exploration borings converted to water wells or from mine water. Problems may exist with such water being contaminated, as has been documented in the Grants Mineral Belt (EPA75), but we do not believe seepage from settling ponds to be a factor.

Infiltration of water discharged to ephemeral stream courses was not calculated separately. It was combined into a lumped term incorporating infiltration and evaporation. Both losses are, in part, a function of surface area. Infiltration takes place primarily in the basin. When three mines are operating, 22.7 km of perennial stream is created and extends into a portion of the basin. Infiltration of the mine effluent adds primarily to the amount of water in storage in the alluvium, versus acting as a source of recharge to the deeper, consolidated strata.

As with many of the intermontane basins in Wyoming, water in the South Powder River Basin is primarily groundwater recharged by sporadic runoff from limited precipitation (Ke77). Some stock ponds that collect surface runoff are supplemented by groundwater from wells or springs. Mine water discharged from one underground mine is used to irrigate approximately 65 hectares of native grass, alfalfa, oats, and barley. In general, groundwater is not used for irrigation (Ho73). Groundwater use for domestic supplies is largely confined to the Dry Fork of the Cheyenne River (Ke77). The number of wells is close to a density of one per 400 ha (Ke77). Typical wells are completed in the alluvium and yield less than 100ℓ /min.

Geological formations in the southern portion of the Powder River Basin include in descending order and increasing age; the 1) Alluvium, 2) Wasatch Formation, 3) Fort Union Formation, 4) Lance Formation, 5) Fox Hills Formation, and 6) older rocks too deep to be affected by uranium mining (NRC78c). Table 3.28 shows the well depth for each formation, anticipated well yields, and the total dissolved solids content in the vicinity of an active uranium mining and milling project in the South Powder River Basin.

Water quality in the Wasatch and Fort Union Formations ranges widely and appears to correlate with the permeability of the water-bearing sand and proximity to outcrops. No relation of water quality to depth is apparent. Analyses of water from Cenozoic rocks show dissolved solids ranging from less than 100 to more than 8000 mg/ℓ (Ho73). Of the 258 analyses performed by the USGS, 55 showed dissolved solids less than 500 mg/ℓ, 133 less than 1000 mg/ℓ, and 125 more than 1000 mg/ℓ. Sodium, sulfate, and bicarbonate are the dominant ions, and water is usually excessively hard. Iron is characteristically a problem in water from the Wasatch and Fort Union Formations (Ho73). Element distributions show considerable variability due to clay lenses in the sandy units (NRC78c). The clays act as barriers to groundwater movement and preferentially concentrate some elements. Table 3.29 shows the ambient groundwater quality in the immediate area of three active mills in the South Powder River Basin.

In the Wyoming model mine sub-basin, total inflow equals 9 m^3/min or 4.73 x 10^6 m^3/yr, and total annual infiltration loss equals 4.65 x 10^6 m^3 (calculated in Appendix H). Restated, 98.2 percent of the discharge infiltrates and the remainder evaporates.

Infiltration of 4.65 x 10^6 m^3/yr is not likely to continue for the full duration of mining unless the bedrock strata have the same or similar permeability as the alluvium and (or) there is an extensive zone of unsaturated alluvium to provide storage. The alluvium in the Wyoming study area is concentrated along the stream axes, is relatively thin, and is underlain by less permeable bedrock strata. It is probable that a zone of saturated alluvium will gradually develop and extend downstream as mine discharge continues. Recharge from the alluvium to the underlying Wasatch or Fort Union Formations will occur but at a low rate compared to infiltration. Water quality in the alluvium is highly variable (Table 3.29); it may or may not be affected by mine drainage. Adverse impacts, if any, are likely to be a result of uranium, sulfate, and mobile elements.

Table 3.28 Northeastern Wyoming groundwater sources

Geologic Period	Aquifer	Depth Range of Wells, m	Anticipated Well Yield, ℓpm		Total Dissolved Solids, mg/ℓ
			Common	High	
Quaternary	Alluvium	3-30	20-945	1140-2270	106-7340
Tertiary	Wasatch	12-300	4-150	380-2370	160-6620
	Fort Union	45-180	4-110	380	484-3250
Cretaceous	Lance	45-365	4-190	1900	450-3060
	Fox Hills	210-700	75-260	760-1900	1240-3290
	Mesaverde	12-915	57-150	225-265	550-1360
	Cody	30-335	4-20	380-760	6392-12,380
	Frontier	20-610	4-20	380-1135	390-2360
	Dakota	75-1830	95-380	760-3410	218-1820
Jurassic	Sundance	120-210	4-20	95	894-2310
Triassic	Spearfish	6-275+	4-115	380-760	2590
Pennsylvanian	Minnelusa	75-1980	95-950	1860-7470	255-3620
Mississippian	Pahasapa	150-2320	380-9460	26,500-35,600	290-3290
Ordovician	Bighorn	0-60	3785	3785	427-3219
Cambrian	Flathead	20-1800	760		124

Source: NRC78b.

Table 3.29 Groundwater quality of wells sampled by the three major uranium producers in the South Powder River Basin, Wyoming

Parameter	Range of Concentration Reported (mg/ℓ)		
	Kerr-McGee(a)	TVA(b)	Exxon(c)
pH	7.4-8.0	7.4 - 8.5	7.3-8.1
Spec. cond. μmhos/cm	210-1100	250-1300	290-600
Ca	28-343	10-200	26-150
Mg	8-81	2-80	1-13
Na	5-71	10-300	54-121
HCO_3	30-380	70-110	90-412
SO_4	28-980	8-1000	58-575
Cl	<5-57	11-25	6-16
Zn	0.006-18.0	0.03 -3	ND- 0.14(d)
Fe		0.2 -20	0.01- 1.64
Ba			ND- 0.05
Radium (pCi/ℓ)	0.41 - 5.18	0.2 -18	0.4 -12.0
Uranium (mg/ℓ)	< 0.002- 2.3	0.002-60	0.0004 - 0.21

(a) Shallow wells up to 61 meters depth, Tables 2.6-7 through 2.6-10 of reference Ke77.

(b) From Figs. C1 and C3 of reference NRC78b.

(c) Table 2.12 of reference NRC78d.

(d) ND: Not detectable.

An actual example of this saturated front developing and moving downgradient is present at the Kerr-McGee Nuclear Corporation's Bill Smith Mine in South Powder River Basin (Ke77). The mine discharges to a tributary of Sage Creek at a rate of about 1.7 m^3/min. From the period January 1974 to late 1976, a flow front 23 km long developed as a result of infiltration into the sandy alluvium. The discharge water maintains a high groundwater level in the stream bed. Unfortunately, no information is available on the geometry of the stream channel to evaluate the volume of water that has infiltrated in the three-year period or on any water quality changes that have occurred.

In summary, additional field data are needed to properly address the water quality effects of infiltration. Both theory and at least one field example indicate extensive infiltration of effluent containing at least some mobile stable and radioactive contaminants. Therefore, we recommend additional field investigations to determine, at the minimum, any hydraulic and water quality effects of mine discharge on shallow aquifers and the influence of dewatering on regional water levels and water quality, regardless of preexisting or anticipated local water use patterns.

3.3.4 Gases and Dusts from Mining Activities

Dusts and toxic gases are generated from routine mining operations. Combustion products are produced by large diesel and gasoline-powered equipment in the mine and by trucks transporting the overburden, ore, and sub-ore from the pit to storage pile areas. Dusts are produced by blasting, breaking, loading, and unloading rock and ore and by haulage trucks moving along dirt roads. Finally, Rn-222 will emanate from exposed ore in the pit and from the ore as it is broken, loaded, and unloaded. These sources will be discussed individually.

3.3.4.1 Dusts and Fumes

Most vehicular emissions are from the combustion of hydrocarbon fuels in heavy-duty, diesel-powered mining equipment. Surface mines produce considerably more emissions than underground mines, since the overburden must be removed before the ore can be mined. The principal emissions are particulates, sulfur oxides, carbon monoxide, nitrogen oxides, and hydrocarbons. The quantity of these combustion products released to the atmosphere depends on the number, size, and types of equipment used.

The EPA estimates the following emissions from mining 1350 MT of ore per day from a surface mine (Re76).

	Emissions per Operating Day, kg/d	
Pollutant	Mining Operations	Overburden Removal
Particulates	17.0	18.9
Sulfur oxides	35.4	39.3
Carbon monoxide	294.2	327.4
Nitrogen oxides	484.6	538.4
Hydrocarbons	48.4	53.8

Assuming a 330 operating-day-year (Ni79), we adjusted these emission rates to ore production for the average surface mine (1.2×10^5 MT/yr) and the average large surface mine (5.1×10^5 MT/yr) as described in Sections 1.3.1 and 3.3.1. Table 3.30 shows the total airborne combustion product emissions. These estimated emission rates are somewhat higher than rates previously suggested by the U.S. Atomic Energy Commission (AEC74).

Table 3.30 Estimated air pollutant emissions from heavy-duty equipment at surface mines

	Emissions, MT/yr(a)	
Pollutant	Average Mine(b)	Average Large Mine(c)
Particulates	3	14
Sulfur oxides	7	28
Carbon monoxide	55	235
Nitrogen oxides	91	387
Hydrocarbons	9	39

(a)Based on (Re76) and 330 operating days per year (Ni79).

(b)Ore production = 1.2×10^5 MT/yr.

(c)Ore production = 5.1×10^5 MT/yr.

Dust is produced from blasting, scraping, loading, transporting, and dumping ore, sub-ore, and overburden. Additional dust is produced when the ore is reloaded from the stockpile for transportation to the mill. Dust emissions vary widely, depending upon moisture content, amount of fines, number and types of equipment operating, and climatic conditions. Because ore is usually wet, the relative amounts of dust produced from mining and handling it are usually small. We selected the following emission factors from those suggested by the EPA for the above listed mining activities (Hu76, Ra78, Da79):

Blasting = 5×10^{-4} kg dust/MT
Scraping and bulldozing = 8.5×10^{-3} kg dust/MT
Truck loading = 2.5×10^{-2} kg dust/MT
Truck dumping = 2×10^{-2} kg dust/MT

We applied these emission factors to the ore, sub-ore, and overburden production rates of the average mine and average large mine and estimated average annual dust emissions for these mining activities (see Table 3.31). These are probably maximum emission rates because blasting is not always required, and some emission factors appear to have been based upon data from crushed rock operations, which would contain more fines than rock removed from surface mines. One-half the emission factor values were applied to ore and sub-ore because they are usually wet, except when reloading ore from the stockpile, in which case it is assumed to have dried during the 41-day residence period (Section 3.3.1.2).

The movement of heavy-duty haul trucks is probably the largest single source of dust emissions at surface mines. An emission factor (EF) for this source can be computed by the following equation (EPA77b).

$$EF = 2.28 \times 10^{-4} \, (s) \left(\frac{V}{48}\right) \frac{365-W}{365} \, (TF) \, (f) \qquad (3.2)$$

where,

EF = Emission factor, MT/vehicle kilometer traveled (MT/VKmt),
S = Silt content of road surface, percent,

V = Vehicle velocity, kmph [Note: This term becomes $\left(\frac{V}{48}\right)^2$ for velocities less than 48 km/hr (EPA77b, DA79)],

W = Mean annual number of days with 0.254 mm or more rainfall,

TF = Wheel correction factor, and

f = Average fraction of emitted particles in the <30 μm diameter suspended particle size range; particles having diameters greater than 30 μm will settle rapidly near the roadway.

Values selected for these terms in the solution of Equation 3.2 are --

S = 10 percent (Da79),

V = 32 km/hr for heavy-duty vehicles and 48 km/hr for light vehicles (therefore, the velocity term is $(32/48)^2$ and (48/48), respectively),

W = 90 days (EPA77b),

TF = 2.5 (Da79) (heavy-duty vehicles only), and

f = 0.60, since the weight percent of particles of less than 30 μm and greater than 30 μm in diameter is generally considered to be 60 and 40 percent, respectively (EPA77b).

Substituting these values into Equation 3.2 yields 1.15×10^{-3} MT/VKmT and 1.03×10^{-3} MT/VKmt for the emission factors of heavy-duty haul trucks and light duty vehicles, respectively.

Table 3.31 shows estimated dust emissions for the movement of heavy-duty haul trucks using the following information:

Table 3.31 Average annual dust emissions from mining activities

Mining Activity	Dust Emissions, MT/yr					
	Average Mine[a]			Average Large Mine[b]		
	Ore[c]	Sub-ore[c]	Overburden	Ore[c]	Sub-ore[c]	Overburden
Blasting	0.03	0.03	3.0	0.13	0.13	20
Scraping/bulldozing	NA[d]	NA	51	NA	NA	340
Truck Loading	1.5	1.5	150	6.4	6.4	1000
Total at Pit Site	1.53	1.53	204	6.53	6.53	1360
Truck Dumping	1.2	1.2	120	5.1	5.1	800
Reloading stockpiled ore[e]	3.0	NA	NA	13	NA	NA
Total at Pile Sites	4.2	1.2	120	18.1	5.1	800
Vehicular dust[f]	14	14	304	59	59	2020
Wind suspended dust from storage piles	10	3	30	44	10	94

(a) Based on annual production rates of 1.2×10^5 MT of ore and sub-ore and 6.0×10^6 MT of overburden.

(b) Based on annual production rates of 5.1×10^5 MT of ore and sub-ore and 4.0×10^7 MT of overburden.

(c) Assumed wet.

(d) NA - not applicable.

(e) Assumed dry.

(f) Dust emissions from heavy-duty vehicular traffic along ore, sub-ore and overburden haul roads.

EF = 1.15×10^{-3} MT/VKmt,

Truck capacities = 31.8 MT for ore and sub-ore and
109.1 MT for overburden (Da79),

Round-trip haul distance = 3.2 km to ore and sub-ore piles
and 4.8 km to overburden dump, and

Annual production rates = given in Section 3.3.1 and in the
footnotes of Table 3.31.

Additional dust emissions will occur from the movement of light-duty vehicles along access roads. Using the emission factor derived above (1.03×10^{-3} MT/VKmT) and assuming that there are 24 km of access roads traveled 4 times a day for 330 operating days per year, about 33 MT of dust will be produced from this source annually. Emissions during haulage road maintenance is relatively small and will not be considered.

Table 3.31 also shows average annual dust emissions from wind erosion of overburden, sub-ore, and ore piles at the model surface mines. For these computations, we assumed the model overburden pile to be that of Case 2 and in the shape of a 65-m high truncated cone (Table 3.11). The same was assumed for the average mine, except the pile height was 30 m. The sub-ore piles of both mines were assumed to have a truncated cone configuration (Table 3.20). The same configuration was also assumed for the ore piles, but the pile heights were 9.2 m for the average large mine and 3.1 m for the average mine (Table 3.17).

Emission factors, computed in Appendix I, are 0.850 MT/hectare-yr for overburden and sub-ore piles and 0.086 kg/MT for the ore stockpiles. The first emission factor was multiplied by the overburden and average sub-ore pile areas; the second factor was multiplied by the annual ore production.

In computing the Table 3.31 dust emissions, we assumed no effective dust control program and that there was no vegetation on overburden and sub-ore piles. Haul roads are normally sprinkled routinely during dry periods, and stabilizing chemicals are applied primarily to ore haul roadways at some mines. Sprinkling can reduce dust emissions along haul roads by 50 percent, and up to 85 percent by applying stabilizing chemicals (EPA77b, Da79).

The dust emissions from vehicular traffic (Table 3.31) (transportation) were summed with those produced by light vehicular traffic (33 MT/yr) and considered as one source of emissions. Concentrations of contaminants in the dust are unknown. Some spillage of ore and sub-ore along haul roads will undoubtedly raise uranium levels in roadbed dust. As an estimate, uranium and daughter concentrations in the dust were considered to be twice background, 8 ppm (2.7 pCi/g), while concentrations of all other contaminants were considered to be similar to those in overburden rock (Section 3.3.1.1, Table 3.16). Table 3.32 shows the annual emissions computed with these assumptions.

Table 3.33 lists annual contaminant emissions from mining activities (scraping, loading, dumping, etc.) according to source location, at the pit and at the piles. Contaminant emissions were computed by multiplying the total annual dust emissions at each pile (Table 3.31) by the respective contaminant concentrations in each source -- overburden (Section 3.3.1.1.; Table 3.16), sub-ore (Section 3.3.1.3; Table 3.19) and ore (Section 3.3.1.2; Table 3.19). Contaminant emissions at the site of the pit were computed by multiplying the total annual dust emissions of ore, sub-ore, and overburden (Table 3.31) by their respective contaminant concentrations. The three products of the multiplication were then summed to give the values in the 4th and 8th data columns of Table 3.33. The health impact of the sources at each location will be assessed separately in Section 6.1.

Table 3.34 lists annual contaminant emissions due to wind suspension and transport of dust. These values were computed by multiplying the annual mass emissions (Table 3.31) by the contaminant concentrations in overburden, sub-ore, and ore listed in Sections 3.3.1.1, 3.3.1.3, and 3.3.1.2, respectively. The uranium and uranium daughter concentrations were also multiplied by an activity ratio (dust/source) of 2.5 (Section 3.3.1.2). Although some metals may also be present as secondary deposits, it was believed that there were insufficient data to justify multiplying their concentrations by the 2.5 ratio.

3.3.4.2 Radon-222 from the Pit, Storage Piles, and Ore Handling

Rn-222 will be released from the following sources during surface mining operations:

Table 3.32 Average annual emissions of radionuclides (μCi) and stable elements (Kg) from vehicular dust at the model surface mines

Contaminant	Average Large Surface Mine[a]	Average Surface Mine[b]
Arsenic	20	3.3
Barium	630	106
Copper	39	6.6
Chromium	<111	<19
Iron	13,030	2,190
Mercury	<17	<2.9
Potassium	15,200	2,560
Manganese	1,050	177
Molybdenum	5.4	0.9
Lead	48	8.0
Selenium	4.3	0.7
Strontium	330	55
Vanadium	220	37
Zinc	43	7.3
Uranium-238 and each daughter	5,860	990
Thorium-232 and each daughter	2,170	370

(a) Mass emissions = 2,170 MT/yr.

(b) Mass emissions = 365 MT/yr.

Table 3.33 Average annual emissions of radionuclides (μCi) and stable elements (kg) from mining activities at the model surface mines

	Average Surface Mine[(a)]				Average Large Surface Mine[(a)]			
Contaminant	Overburden Pile Site	Sub-ore Pile Site	Ore Pile Site	Pit Site	Overburden Pile Site	Sub-Ore Pile Site	Ore Pile Site	Pit Site
Arsenic	1.1	0.10	0.36	2.1	7.2	0.44	1.6	13
Barium	35	1.1	3.9	62	232	4.7	17	406
Cobalt	NR[(b)]	0.02	0.07	0.05	NR	0.08	0.29	0.21
Copper	2.2	0.07	0.26	3.9	14	0.31	1.1	25
Chromium	<6	0.02	0.08	<10	<41	0.10	0.36	<70
Iron	720	19	66	1,270	4,800	80	284	8,360
Lead	2.6	0.09	0.33	4.7	18	0.40	1.4	31
Magnesium	NR	4.2	15	11	NR	18	63	46
Manganese	58	1.2	4.0	102	388	4.9	17	672
Mercury	<1	ND[(c)]	ND	<1.6	<6.4	ND	ND	<11
Molybdenum	0.3	0.14	0.48	0.86	2.0	0.59	2.1	4.9
Nickel	NR	0.02	0.08	0.06	NR	0.10	0.36	0.26
Potassium	840	30	105	1,500	5,600	128	453	9,850
Selenium	0.2	0.13	0.46	0.74	1.6	0.56	2.0	4.2
Strontium	18	0.16	0.55	31	120	0.66	2.4	206
Vanadium	12	1.7	5.9	25	80	7.2	26	154
Zinc	2.4	0.04	0.12	4.2	16	0.15	0.52	28
Uranium-238 & each daughter	1,800	120	2,990	4,300	12,000	510	12,900	25,700
Thorium-232 & each daughter	120	2.4	42	220	800	10	180	1,440

(a) Mass emissions from Table 3.31.

(b) NR - Not reported.

(c) ND - Not detected.

Table 3.34 Average annual emissions of radionuclides (μCi) and stable elements (kg) in wind suspended dust at the model surface mines

	Average Large Surface Mine			Average Surface Mine		
Contaminant	Overburden Pile	Sub-Ore Pile	Ore Stockpile	Overburden Pile	Sub-Ore Pile	Ore Stockpile
Arsenic	0.85	0.86	3.8	0.27	0.26	0.86
Barium	27	9.2	40	8.7	2.8	9.2
Cobalt	NR[a]	0.16	0.70	NR	0.05	0.16
Copper	1.7	0.61	2.7	0.54	0.18	0.61
Chromium	<4.8	0.20	0.88	<1.5	0.06	0.20
Iron	564	157	690	180	47	157
Mercury	<0.75	ND[b]	ND	<0.24	ND	ND
Potassium	660	250	1,100	210	75	250
Magnesium	NR	35	154	NR	11	35
Manganese	46	9.6	42	15	2.9	9.6
Molybdenum	0.24	1.2	5.0	0.08	0.35	1.2
Nickel	NR	0.20	0.88	NR	0.06	0.20
Lead	2.1	0.78	3.4	0.66	0.23	0.78
Selenium	0.19	1.1	4.8	0.06	0.33	1.1
Strontium	14	1.3	5.7	4.5	0.39	1.3
Vanadium	9.4	14	62	3.0	4.2	14
Zinc	1.9	0.29	1.3	0.60	0.09	0.29
Uranium-238 & each daughter	1,410	1,000	31,300	450	300	7,100
Thorium-232 & each daughter	94	20	440	30	6.0	100

[a] NR - Not reported.

[b] ND - Not detected.

1. Ore, sub-ore, and overburden during rock breakage and loading in the pit and unloading on the respective piles. (Since rock breakage, loading, transporting, and unloading usually occur in a short time period, they are considered one release.)
2. Ore during reloading from the stockpile after a 41-day residence time (Section 3.3.1.2).
3. Exposed surfaces of overburden, ore, and sub-ore in the active pit area.
4. Overburden, ore, and sub-ore pile surfaces.

The annual quantities of Rn-222 released from sources 1 and 2 above were computed using the following factors and assumptions:

1. Rn-222 is in secular equilibrium with U-238.
2. The density of ore, sub-ore, and overburden is 2.0 MT/m^3.
3. Annual production rates of ore, sub-ore, and overburden are those given previously in this Section and in footnotes "a" and "b" of Table 3.31.
4. All Rn-222 present, 0.00565 Ci/m^3 per percent U_3O_8, is available with an emanation coefficient of 0.27. [Although an emanation coefficient of 0.2 is commonly used (Ni79), recent emanation-coefficient measurements for 950 samples of domestic uranium ores by the Bureau of Mines indicate a value between 0.25 and 0.3 to be more appropriate (Au78, Tanner, A.B., Department of Interior, Geological Survey, Reston, VA, 11/79, personal communication). Therefore, an emanation coefficient of 0.27 was selected.]
5. The quantities of U_3O_8 present in ore, sub-ore, and overburden are 0.10, 0.015, and 0.0020 percent, respectively.

Substituting these values into the following equation yields the Rn-222 releases given in Table 3.35 for the average mine and the average large mine.

$$\text{Rn-222 (Ci/yr)} = (\text{Percent } U_3O_8)\left(\frac{0.00565 \quad \text{Ci}}{m^3 \text{ x percent}}\right)(0.27)\left(\frac{M^3}{2.0 \text{ MT}}\right) \qquad (3.3)$$

$$\text{X (Production Rate, } \frac{MT}{yr})$$

The quantities of Rn-222 that emanate from exposed overburden, ore, and sub-ore surfaces in the pit were estimated by the following method. Exposed surface areas of ore and sub-ore are assumed equal since equal quantities of each are mined. The computation assumes an ore plus sub-ore zone 12 m thick (h_1) in the shape of a truncated cone with 45 degree sloping sides (Fig. 3.14). The radii of the zone, r_1 and r_2, can be computed using the following equation from the relationship $r_2 = r_1 + 12$ and the volumes of ore plus sub-ore mined in a 2.4 year period -- $1.22 \times 10^6 m^3$ and $2.8 \times 10^5 m^3$ at the average large mine and average mine, respectively (the bulking factor is not considered in computing the pit volume).

$$V \text{ (ore + sub-ore zone)} = 1/3\ \pi h_1\ (r_1^2 + r_1 r_2 + r_2^2) \qquad (3.4)$$

The computed radii, r_1 and r_2, were 174 m and 186 m at the average large mine and 80 m and 92 m at the average mine. The surface areas (S_A) of exposed ore and sub-ore in the pit are then one-half that given by the equation,

$$S_A = 1/2\pi\ (d_1 + d_2)(\text{slant height}) + \pi r_1^2, \qquad (3.5)$$

where d_1 and d_2 are the diameters related to r_1 and r_2. Exposed surface areas of ore and sub-ore were computed to be equal and 57,170 m^2 at the average large mine and 14,650 m^2 at the average mine.

The shape of the overburden zone was assumed to be the same as the ore and sub-ore zone (Fig. 3.14). The thickness, h_2, and radius, r_3, of this zone can be computed using the following equation with the relationship, $r_3 = r_2 + h_2$, and knowing the volume--$4.8 \times 10^7 m^3$ and $7.2 \times 10^6 m^3$--at the average large mine and average mine, respectively.

$$V \text{ (overburden)} = 1/3\ \pi h_2\ (r_2^2 + r_2 r_3 + r_3^2) \qquad (3.6)$$

Since r_2 was computed above to be 186 m at the average large mine and 92 m at the average mine, Equation 3.6 becomes

$$4.8 \times 10^7 = 1.087 \times 10^5 h_2 + 584 h_2^2 + 1.047 h_2^3 \qquad (3.7)$$

for the average large mine, and

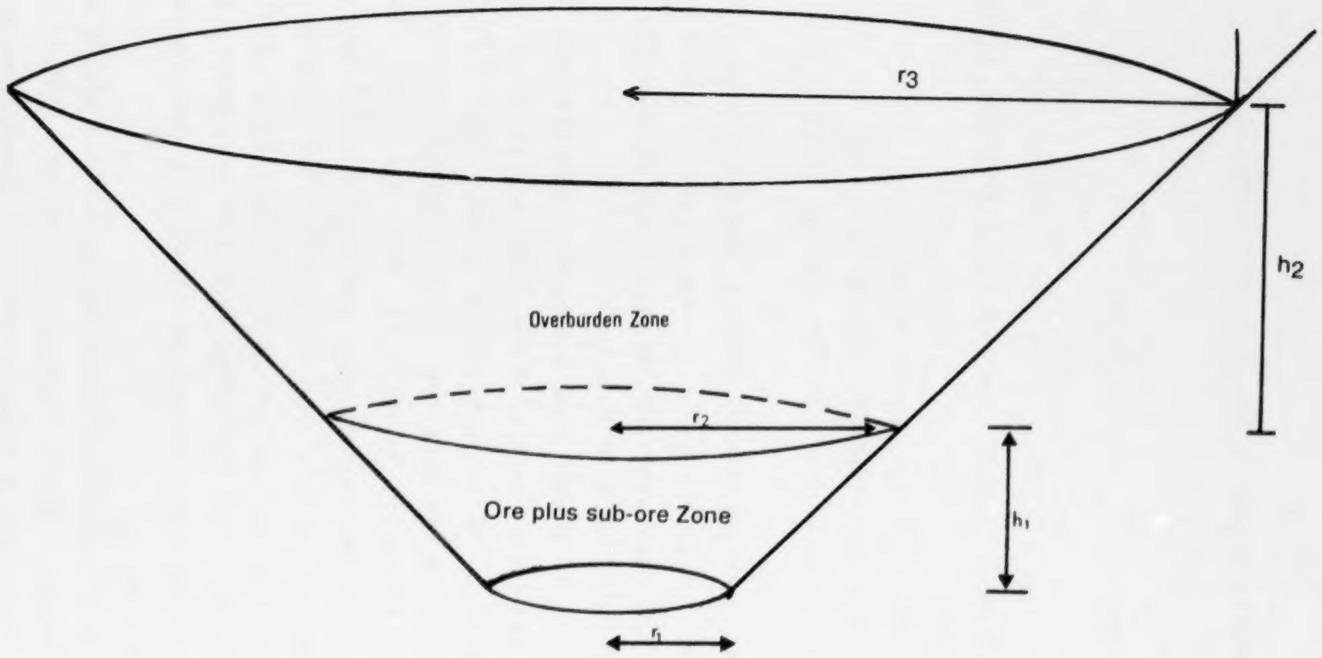

Figure 3.14 Configuration of open pit model mines.

$$7.2 \times 10^6 = 2.659 \times 10^4 h_2 + 289\ h_2^{\ 2} + 1.047 h_2^{\ 3} \qquad (3.8)$$

for the average mine.

Solving these equations yields the following parameters:

	h_2	r_3	r_2
average large mine	188 m	374 m	186 m
average mine	105 m	197 m	92 m

The surface area (S_A) of the exposed overburden is then given by the following equation.

$$S_A = 1/2\,\pi\,(d_2 + d_3)\ \text{(slant height)}, \qquad (3.9)$$

where d_1 and d_2 are the diameters related to r_2 and r_3. Areas computed were $4.68 \times 10^5 m^2$ and $1.34 \times 10^5 m^2$ for the average large mine and average mine, respectively.

Multiplying the exposed ore, sub-ore, and overburden areas by their U_3O_8 contents (0.10%, 0.015% and 0.002%, respectively) and by a Rn-222 exhalation rate of 0.092 Ci/m^2 per year per percent U_3O_8* and summing gives the annual Rn-222 releases shown in Table 3.35.

The emanation of Rn-222 from overburden, sub-ore, and ore storage piles is based on an exhalation rate of 0.092 Ci/m^2 per yr per percent U_3O_8 (Ni79), and ore grades of 0.002 percent, 0.015 percent, and 0.10 percent, respectively. The surface areas used were those computed previously for the case 2 model mines and listed in Tables 3.11, 3.17 and 3.20. The areas for the average large mine and average mine are $1.1 \times 10^6 m^2$ and $2.2 \times 10^5 m^2$ for overburden piles, $1.2 \times 10^5 m^2$ and $3.6 \times 10^4 m^2$ for sub-ore piles, and $6.2 \times 10^3 m^2$ and $3.6 \times 10^3 m^2$ for the ore piles, respectively. Applying these parameters, the annual Rn-222 emissions from the overburden, sub-ore, and ore piles at the average mine and average large mine were computed. Table 3.35 presents the results.

The total annual Rn-222 released during surface mining operations is the sum of the releases from the sources considered: 331 Ci from the average mine and 1261 Ci from the average large mine. Considering ore production and

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*The average value of measured exhalation rates at surface uranium mines (Ni79).

Table 3.35 Radon-222 releases during surface mining, Ci/yr

Source	Average Mine	Average Large Mine
Ore loading and unloading	9	39
Reloading ore from stockpile	9	39
Sub-ore loading and unloading	1	6
Overburden loading and unloading	9	61
Exposed surface of overburden, ore, and sub-ore in the pit	180	691
Ore stockpile exhalation	33	57
Sub-ore pile exhalation	50	166
Overburden pile exhalation	40	202
Total	331	1261

grade differences, these values agree reasonably well with those computed by other procedures (Tr79).

3.4 Underground Mining

3.4.1 Solid Wastes

During underground mining, like surface mining, materials are removed, separated according to ore content, and stored on the surface for various periods of time (Section 1.3.3). These separate piles consist of waste rock produced from shaft sinking operations and from cutting inclines, declines, and haulage drifts through barren rock, sub-ore, and ore. The waste rock is similar to overburden removed at surface mines, except much smaller quantities are involved and none are returned to the mine. However, as mining progresses, waste rock is sometimes used to backfill mined out areas of the mine and retained beneath the surface. The ore and sub-ore will also be similar in nature to those described previously for surface mines, as is their potential to be sources of contamination to the environment (Fig. 3.15).

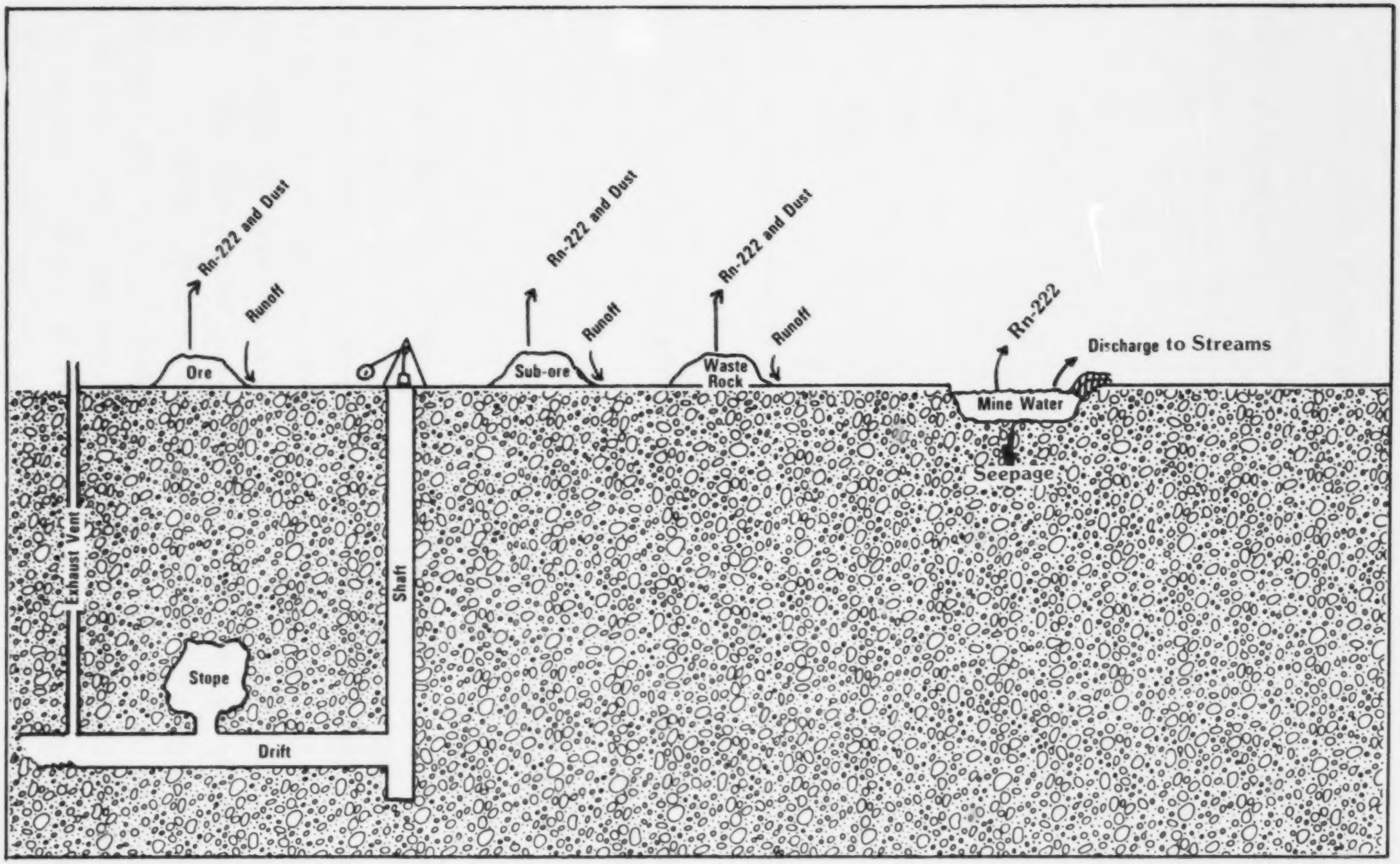


Figure 3.15 Potential sources of environmental contamination from active underground uranium mines.

3.4.1.1 Waste Rock Piles

Much smaller quantities of waste rock accumulate at underground mines than overburden at surface mines. The weight ratio of waste rock to ore depends mainly upon the size, depth, and age of the mine. During the initial mining stages, all material removed is waste rock. As entry into the ore body occurs and ore mining begins, the quantity of waste rock removed per metric ton of ore decreases sizably. Once in the ore body, as little waste rock as possible is mined. The ratio of ore to waste rock removed from underground mines varies considerably. At seven presently active underground mines, the ore to waste rock ratio varies from 1.5:1 to 16:1, with an average ratio of 9.1:1 (Jackson, P.O., Battelle Pacific Northwest Laboratory, Richland, WA, 12/79, personal communication). As future mines become larger and deeper, the overall ore to waste rock ratio will probably decrease.

Since the annual average ore capacity of underground mines was 1.8×10^4 MT in 1978 (Section 1.3.1), the average of the 305 underground mines would have produced 2.0×10^3 MT of waste rock during that year, assuming the average 9.1:1 ore to waste rock ratio. This will be considered the production rate of the "average underground mine." Like surface mines, relatively few of the 305 active underground mines account for a significant portion of the total ore produced by the underground method. Also, future underground mines are expected to have larger capacities than many of the current mines (Th79). Therefore, a second underground mine will be considered, which is defined as the "average large mine." Its annual ore production rate is assumed to be 2×10^5 MT, the average ore capacity of five large underground operations (Ja79b, TVA79, TVA76, TVA78a, TVA78b). The quantity of waste rock removed annually will be 2.2×10^4 MT, assuming the ore to waste rock ratio to be the same as for the average mine. Assuming the density of waste rock to be about 2.0 MT/m^3 and a bulking factor of 1.25 (Burris, E., Navajo Engineering Construction Authority, Shiprock, N.M., 2/80, personal communication), the average mine and average large mine will produce waste rock at an annual rate of 1.3×10^3 m^3 and 1.4×10^4 m^3, respectively. Since waste rock is not presently used to backfill mined-out areas, this rate of accumulation will continue for the life of the mine, which is assumed to be the same as that for an open pit mine, 17 years.

Table 3.36 lists estimated average surface areas of the waste rock piles during the lifetimes of the two mines defined above. The following parameters were used:

Parameter	Average Mine	Average Large Mine
Waste rock production rate, MT/yr	2.0×10^3	2.2×10^4
Rock density, MT/m^3	2.0	2.0
Bulking factor	1.25	1.25
Waste rock volume, m^3/yr	1.3×10^3	1.4×10^4
Active mine life, yr	17	17
Pile height, m	6	12

These estimated areas assume no backfilling and that the piles are on level terrain. Because waste rock is sometimes used to backfill and is often dumped into a gorge or ravine, these surface areas represent maximum conditions.

The mineralogy, physical characteristics, and composition of waste rock from underground mines are assumed to be identical to the overburden removed from open pit mines (Section 3.3.1.1). Also, reclamation procedures for waste rock piles at underground mines should be similar to those described in Section 3.3.1.4 for overburden dumps.

3.4.1.2 Ore Stockpiles

Because ore is often stockpiled at the mine and/or at the mill, it becomes a potential source of contamination to the mine environment during the storage period. These piles will be smaller than the waste rock piles, but the concentration of most contaminants in the ore-bearing rock will be much greater.

Ore stockpile residence times can vary considerably with time and ore management. Residence times commonly range from a few days to a few months. The same residence time will be assumed for underground mines as was selected above for surface mines, 41 days. Assuming a 330 operating-day-year and a 1.25 bulking factor, the ore stockpiles of the average mine and average large mine will contain 1,400 m^3 and 15,500 m^3 of ore, respectively. The surface areas of the ore stockpiles were computed using these volumes and assuming 3.1 m high rectangular piles (NRC78a). Table 3.37 lists the estimated surface areas.

Table 3.36 Estimated average surface areas of waste rock piles at underground mines

Mine Size	Average Accumulation,(a) m^3	Surface Area of Pile, m^2	Surface Area of Pad, m^2
Average mine(b)	1.1×10^4	2,700	2,460
Average large mine(c)	1.2×10^5	14,100	12,800

(a)Assumes average volume of waste rock accumulated during 17-yr. mine life with no backfilling (1/2 total volume accumulation).

(b)Annual waste rock production = 2.0×10^3 MT.

(c)Annual waste rock production = 2.2×10^4 MT.

Note.--Waste rock piles are rectangular with length twice the width and sides sloping at 45° (Fig. 3.8 a).

Table 3.37 Estimated surface areas of ore stockpiles at underground mines

Mine Size	Steady State Accumulation,(a) m^3	Surface Area of Pile, m^2	Surface Area of Pad, m^2
Average mine(b)	1,400	680	620
Average large mine(c)	15,500	5,800	5,480

(a)Assume 41-day residence time.

(b)Annual ore production = 1.8×10^4 MT.

(c)Annual ore production = 2×10^5 MT.

Note.--Ore stockpiles are rectangular with length twice the width and sides sloping at 45° (Fig. 3.8 a). Pile height is assumed to be 3.1 m (NRC78a).

The mineralogy, physical characteristics, and composition of ore from underground mines are assumed to be identical to the ore removed from surface mines (Section 3.3.1.2). The U_3O_8 grade of ore may average somewhat higher from underground mines than from surface mines. However, a grade of 0.1 percent U_3O_8 probably approximates reasonably well the ore reserves minable by the underground method (DOE79). Uranium and its decay products in airborne dust from these ore piles will be concentrated by a factor of 2.5 (Section 3.3.1.2).

3.4.1.3 Sub-Ore Piles

The quantity of sub-ore mined at an underground mine, as at a surface mine, is considered to be about equal to the quantity of ore mined, 1.8×10^4 MT at the average mine and 2×10^5 MT at the average large mine. Assuming sub-ore to have a density of 2.0 MT/m^3 and after removal a bulking factor of 1.25, the average volume of sub-ore to be on the surface during the 17-yr operational life of the average mine and average large mine will be 9.6×10^4 m^3 and 1.1×10^6 m^3, respectively (i.e., one-half the total of 17-yr accumulation).

Although sub-ore is often placed on top of piles of previously mined waste rock (Perkins, B.L., New Mexico Energy and Minerals Department, Santa Fe, NM, 12/79, personal communication), we assumed separate rectangular piles in computing the surface areas of the piles at the model mines. Table 3.38 lists the estimated surface and pad areas of the sub-ore piles. These computations were based on pile heights of 6 m at the average mine and 12 m at the average large mine.

At underground mines, the cutoff grade ranges from 0.02 to 0.05 percent U_3O_8, yielding an average sub-ore grade of 0.035 percent U_3O_8 (99 pCi/g) (Perkins, B.L., New Mexico Energy and Minerals Department, Santa Fe, N.M., 12/79, personal communication). The mineralogy, physical characteristics, and other constituents of sub-ore from underground mines are assumed identical to the sub-ore removed from surface mines (Section 3.3.1.3).

Table 3.38 Estimated average surface areas of sub-ore piles at underground mines

Mine Size	Average Accumulation,(a) m^3	Surface Area of Pile, m^2	Surface Area of Pad, m^2
Average mine(b)	9.6×10^4	18,800	17,700
Average large mine(c)	1.1×10^6	104,900	99,400

(a) One-half that which will accumulate during the 17-yr mine life.

(b) Annual sub-ore production = 1.8×10^4 MT.

(c) Annual sub-ore production = 2.0×10^5 MT.

Note.--Sub-ore piles are rectangular with length twice the width and sides sloping at 45° (Fig. 3.8a).

3.4.2 Mine Water Discharge

3.4.2.1 Data Sources

Information concerning the amount and quality of water discharged from underground uranium mines in New Mexico is from field surveys conducted in 1975 (EPA75, P. Frenzel, USGS, written communication, 1979) and Wogman (Wo79), from site-specific environmental impact statements and reports, from NPDES permits, and from a State study (Pe79).

Many mining companies maintain that permits are not required because the formerly ephemeral streams into which discharge occurs are, in effect, a result of the discharges and do not meet the definition of navigable bodies of water. Nevertheless, the companies have applied for permits, together with a request to the courts for a ruling concerning their necessity.

The New Mexico district office of the U.S. Geological Survey (L. Beal, USGS, written communication, 1979) provided discharge rate and volume for the regional drainage systems, namely the Rio San Jose, Rio Puerco (east), and the Rio Grande. We followed procedures developed by the USGS (Bo70) to calculate runoff from ungaged basins.

3.4.2.2 Quality and Quantity of Discharge

To estimate average or typical conditions for mine water discharge, 11 projects in Colorado, New Mexico, and Utah were selected. Table 3.39 shows the summarized flow and water quality data. The center of current domestic underground mining is in the Colorado Plateau and the San Juan Basin. In this area, there is an increasing trend toward underground mining. In Wyoming, both underground and surface mining activity are significant. In Texas, surface mining and, to a lesser extent, in situ leaching are the principal methods used. Climatic and geologic characteristics and land and water use patterns in the Colorado-Utah-New Mexico uranium area are broadly similar; and the Grants Mineral Belt in general and the Ambrosia Lake District in particular are representative of this area. There are many complicating variables such as the geologic and geochemical characteristics of the ore body and host rock. Water-yield and quality associated with mines also vary within the region, as do the size and relative location of the populace. The Grants Mineral Belt scenario is conservative. The mines discharge relatively large amounts of water to streams that are used for irrigation and stock watering and that flow by or through local centers of population.

Table 3.39 shows discharge from selected underground uranium mines in the Colorado Plateau areas of Colorado, New Mexico, and Utah. On the average, discharge is 2.78 m^3/min, with a standard deviation of 4.34 m^3/min. The selected underground mines discharge an amount of water similar to that from the Wyoming surface mines. In the Grants Mineral Belt area, average flow from 28 underground mines is 2.4 m^3/min (J. Dudley, New Mexico Environmental Improvement Division, written communication). Of the 27 active underground mines being dewatered, 17 discharge to the environment at an average rate of 3.2 m^3/min. The remainder are in a closed circuit. That is, their discharge is used as mill feed water. The range for 17 mines is 0.2 to 19 m^3/min. Average discharges from New Mexico underground mines are significantly greater than those from mines in Colorado and Utah, which average 0.68 m^3/min. Most of the ore production in New Mexico has been from mines 200 to 300 meters deep. In recent years, mines have become progressively deeper and involve more dewatering. For example, the Gulf Mount Taylor mine, which is not yet producing ore, discharges 15 m^3/min and will produce ore from a depth of 1,200 meters. Most of the water is now diverted to a nearby ranch for irrigation and stock watering. When the mill goes on line, most of the mine water will be used there.

Of the 16 active mines in the Ambrosia Lake district, 13 discharge to offsite areas at an average rate of approximately 1.6 m^3/min. For modeling and to be conservative, we assumed that 14 active mines are present in the model mine area and that the average discharge rate per mine is 2.0 m^3/min. This is somewhat less than the average condition for the Grants Mineral Belt (3.2 m^3/min) as a whole in terms of discharge rate, but the high density of mines assumed present in the model area partly compensates for the difference.

For the New Mexico project shown in Table 3.39, numbers 4, 5, 6, and 7 have discharge that comes directly from the mine portal to settling ponds before discharge. Neither ion exchange for uranium recovery nor barium chloride treatment for radium removal is used. Facilities 8 through 11 use ion exchange columns for uranium removal before discharge. Settling may or may not be used, depending on the suspended solids content of the particular discharge. Project number 10 removes radium prior to discharge. Radium concentrations in the combined effluent from two active mines in the Churchrock area (projects 8 and 4), both of which use settling ponds as the only treatment, have ranged from 1.9 to 8.9 pCi/ℓ since 1975. In the first survey (EPA75), effluent from these same mines contained 30.8 and 7.9 pCi/ℓ. The combined discharge from both mines was sampled by the U.S. Geological Survey in 1975, 1977, and 1978 (P. Frenzel, written communciation) and by the EPA (EPA75) in 1975. Concentrations were 30, 14, 2.6, and 2.6 pCi/ℓ, respectively.

It is apparent that there are marked temporal trends in mine water quality and quantity. Major factors responsible include changes in the dewatering rate accompanying shaft sinking versus actual ore production. Simultaneously, there are changes in the mineral quality and leaching rate of strata as the ore body is approached and then penetrated. Mining practices, oxidation of the ore body and possibly bacterial action may also assist in the solubilization of toxic stable and radioactive trace elements. Sample handling and analytical procedures can also markedly affect results. For example, if suspended solids are high and a sample is acidified prior to filtering, soluble radium, uranium, and other trace constituents typically

Table 3.39 Summary of average discharge and water quality data for underground uranium mines in the Colorado Plateau Region (Colorado, New Mexico, Utah) and a comparison with NPDES limits

	Discharge	Dissolved Radioactivity			Major and trace constituents, mg/ℓ							
		Total U,	Ra-226,	Pb-210,								
Project	m^3/min	mg/ℓ	pCi/ℓ	pCi/ℓ	TSS	SO_4	Zn	Ba	Cd	As	Mo	Se
Utah												
1[(a)]	0.67	1.35	1.25		7.5							
Colorado												
2	1.31	2.20	0.53		14.3	872	0.02	0.19	< 0.01	< 0.01	0.4	
3	0.06	0.25	10.00		144.9		0.065		0.003	0.055	0.054	
New Mexico												
4	14.67	1.0	89[(b)]	15	25.4	60.6		2.13		<0.005	< 0.01	0.03
5	3.79	0.67	23[(c)]	33	2.6	213.7				0.011	0.24	0.008
6	1.89	0.02	14[(c)]	15	51.5	744				0.005		0.004
7	0.95	0.18	0.1	0		1045				<0.005	0.05	0.002
8[(a)]	0.18	4.2	1.9	9.7		67.2		0.88		<0.005	< 0.01	0.094
9[(a)]	0.82	1.9	4.7	16	1	675		0.17		0.011	0.45	0.407
10[(a)]	6.06	1.1	2.3	14	1.08	705				< 0.005	0.62	0.027
11[(a)]	0.216	2.6	4.3	14	2.2	837		0.66		0.012	0.79	0.036
Average	2.78	1.41	13.7	14.6	27.8	580		0.81		0.012	0.29	0.076
Standard Deviation	4.34	1.25	25.9	9.1	46.9	368		0.80		0.015	0.29	0.137

Table 3.39 (continued)

State	Summary of NPDES permit limits for daily average/daily maximum, mg/ℓ except Ra-226, pCi/ℓ								
	Dissolved Radium-226	Dissolved Uranium	Total Suspended Solids	Total Dissolved Solids	Zinc	Barium	Cadmium	Arsenic	Vanadium
New Mexico	3/10 10/30 Total Radium	2/4	50/150(day)[d] 20/30(month)		0.5/1.0				
Utah	3/10 and -/3	2/4 and -/2	20/30[e]	NA/650 3500	0.5/1.0				
Colorado	3/10 and -/3	3/5 and 2/4	20/30	48990/122476[f] kg/day	0.5/1.0	1/2 and -/1	0.05/0.1	1/2 and 0.5/1	5/10

[a] Average discharge rate per mine is shown. Two or more mines constitute the project.

[b] $BaCl_2$ treatment for radium removal faulty; repaired in late 1979.

[c] Values shown are for untreated water. $BaCl_2$ treatment now used.

[d] Applies to discharge associated with shaft construction.

[e] Maximum of 10 mg/ℓ for 30-day period and 20 mg/ℓ for 7-day period effective July 1, 1980.

[f] Receiving water standard.

Source: Chemical analyses from in-house studies (EPA75) and State of New Mexico (J. Dudley, Environmental Improvement Division, written communication). NPDES permit data from Regions VI, VIII (H. May, R. Walline, written communication). Other references include site-specific reports (EIS,ER) and company monitoring data.

will increase, as compared to samples that are filtered prior to acidification (Ka77). Therefore, development of "average" or "typical" trace element concentration data is questionable and may be erroneous without detailed knowledge of the many variables affecting the final results.

Despite the foregoing difficulties, available chemical data assembled in Table 3.39 provide much of the source term input data used in subsequent calculations. The reader should bear in mind that uranium concentrations are likely to be less than 3 mg/ℓ simply because it is economically practical to use ion exchange recovery for concentrations greater than this level. Daily average radium-226 concentrations on the order of 3 pCi/ℓ are specified in valid NPDES permits, and reliable data from USGS, EPA, and state sources reveal stream concentrations near the point of discharge to be on the order of 3 to 14 pCi/ℓ in recent years. Therefore, the "average" radium-226 concentration of 13.7 pCi/ℓ used in the subsequent modeling calculations is at least slightly conservative. Actual concentrations of stable elements (Zn, Ba, Cd, etc.) appear to be well below the NPDES limits, which were also developed from analysis of uranium mine effluent. Thus, it is presumed that the average values in Table 3.39 for these elements are reasonably correct. The variables of mine size, age, host rock, and water treatment (ion exchange, barium chloride, settling ponds) are reflected in the data. Water quality for mines examined in Utah and Colorado generally agrees with the New Mexico cases, with the exception of Project Number 3 mine, which is being dewatered and may, therefore, temporarily have excessive suspended solids. We recommend that the NPDES data for uranium mine discharges be evaluated and that additional compliance monitoring be conducted to confirm the quality of mine discharge. Such studies should focus on situations where mine water is being used for irrigation and stock watering.

Table 3.40 shows discharge and water quality characteristics for underground mines under construction and not yet producing ore. The first example involves water pumped from a deep mine shaft under construction. Considerable water is encountered above the ore body; water quality is good and representative of natural conditions; and suspended solids are high as a result of construction. The second case is similar except that flow is reduced, but radium and suspended solids concentrations are greatly elevated due to construction and possible ore body oxidation. The third case involves

Table 3.40 Water quality associated with underground mines in various stages of construction and operation

Project	Discharge	Dissolved Total U	Dissolved Ra-226	Dissolved Pb-210	Concentration, mg/ℓ TSS	Concentration, mg/ℓ SO_4	Concentration, mg/ℓ As	Concentration, mg/ℓ Mo	Concentration, mg/ℓ Se
	m^3/min	mg/ℓ	pCi/ℓ	pCi/ℓ					
New Mexico									
1. Underground mine shaft construction; dewatering	5.76	0.03	0.07	10	23.8	134	<0.005	0.01	0.003
2. Underground mine shaft construction; dewatering	1.73	<0.01	29	0	554	527	0.012	0.007	0.005
3. Underground mine; dewatering wells	1.43	0.08	0.2	0	1	144	<0.005	0.01	0.003
4. Underground mine recirculating leach solution from stopes (after ion exchange)	0	0.32	29	17	1.1	1060	<0.005	3.2	0.268

Source: J. Dudley, State of New Mexico, written communication, 1979.

dewatering wells used to dewater the ore body before mining. There is no oxidation and suspended solids are very low as is radium-226. Dissolved radium-226 in the ore body is on the order of 10 pCi/ℓ or less in the natural state, but concentrations rise to 100 pCi/ℓ or more after mining takes place, possibly due to oxidation and bacterial action in the workings (EPA75). Project Number 4, in the Ambrosia Lake district, is an inactive underground mine now used as a type of in situ leach facility. Mine water is recirculated through the workings. Leached uranium is selectively recovered using ion exchange. The process is a closed one, hence no effluent is involved. Water quality after uranium removal reflects the buildup in radium, lead-210, sulfate, molybdenum, and selenium.

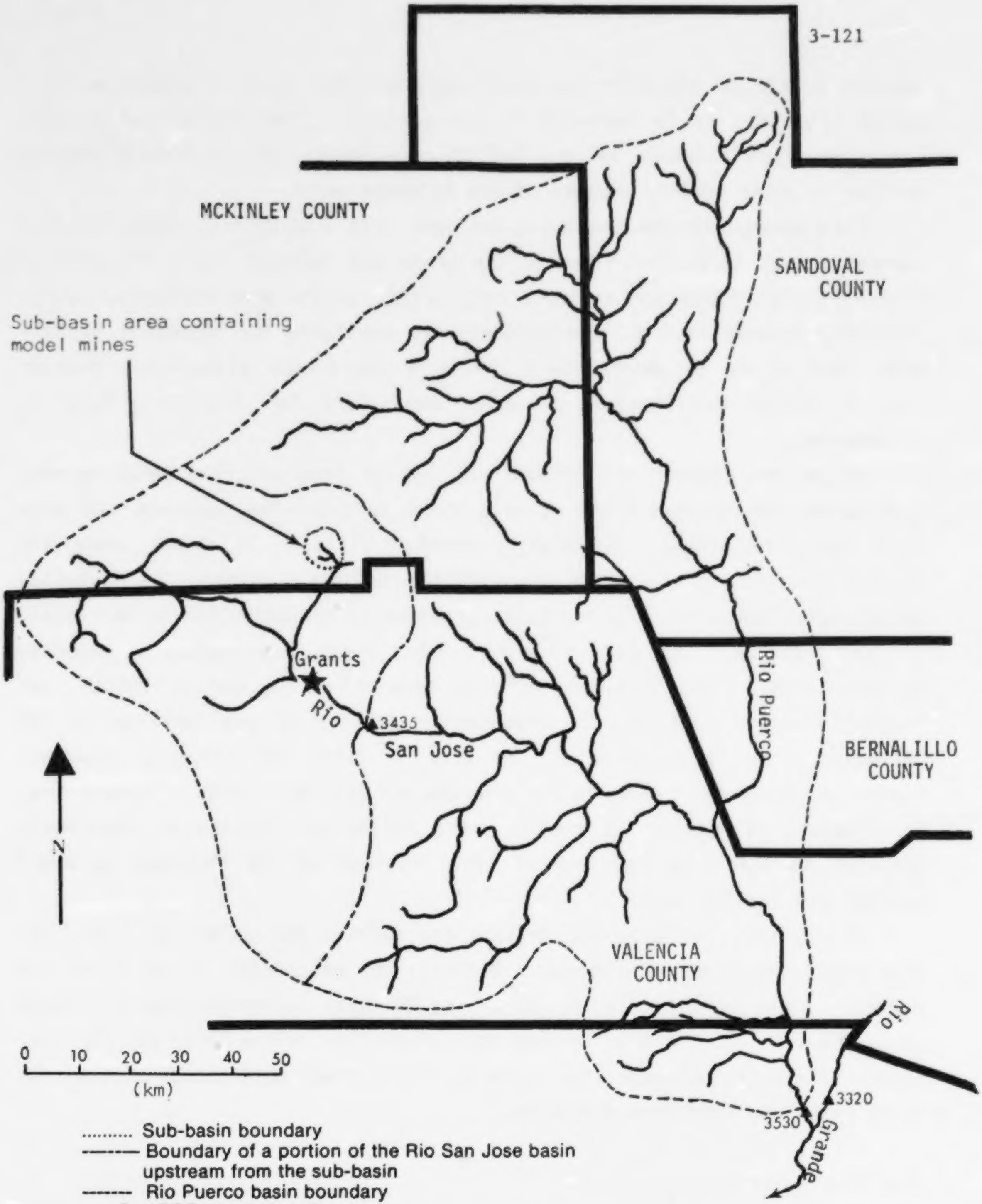
3.4.3. Hydraulic and Water Quality Effects of Underground Mine Discharge

3.4.3.1 Runoff and Flooding in the Model Underground Mine Area

3.4.3.1.1 Study Approach

We chose to study an area of rather concentrated underground mining, similar to the Ambrosia Lake district of New Mexico. All of the mines in the district dewater to different degrees because the principal ore body is in the Westwater Canyon Member of the Morrison Formation, which is also a major aquifer. In the analysis, flows from some 14 active mines discharge to formerly dry washes and dissipate downstream by evaporation and, more importantly, infiltration. Suspended and dissolved constituents persist at the land surface and become available for resuspension and transport in surface floods with recurrence intervals of 2 to 25 years. Contaminated runoff from the sub-basin is then diluted in average annual flows of progressively larger streams and rivers of the region.

Similar to the analysis presented for surface mines in Wyoming, there is a three-basin hierarchy: sub-basin, basin, and regional basin (Fig. 3.16). These correspond to Arroyo del Puerto-San Mateo Creek, Rio San Jose and Rio Puerco, and the Rio Grande. Of these, the Rio Puerco is distinctly ephemeral. The Rio Puerco drains into the Rio Grande, which is perennial, due in large part to the heavily regulated flows and storage reservoirs. Because



.... Sub-basin boundary
------ Boundary of a portion of the Rio San Jose basin upstream from the sub-basin
------- Rio Puerco basin boundary
▲ USGS gauging station
Note: Boundary of the Rio Grande regional basin above Bernardo is not shown.

Figure 3.16 Sketch of sub-basin, basin, and regional basin showing orientation of principal drainage courses, areas of drainage, and location of mines in the New Mexico model area

the Rio Grande is the major regional river and the basis of extensive irrigation projects, it is included in the analysis. The mining area is well away from the Rio Grande Valley, and it is unlikely that noticeable changes in flow or water quality because of mining would occur.

Flow volumes for the sub-basin and open file USGS data (L. Beal, written communication, 1979) for flows in the basin and regional basin are used to transport and dilute contaminants originating in the mine effluent. It is initially assumed that all contaminants are available for transport by surface flow so as to deliberately create a worst-case situation. Section 3.4.3.2 reviews infiltration of water and solute for possible effects on groundwater.

We do not address the effects of seepage from settling ponds because such ponds are relatively small, tend to be self-sealing, and are well away from inhabited areas. Supposedly, settled solids from these ponds are removed and incorporated with uranium mill tailings. Limited field studies to determine whether such ponds cause groundwater contamination is warranted. In some instances, the ponds have synthetic liners, and leakage is expected to be minimal. The influence of mine dewatering (by wells, shafts, and pumping of mine workings) on groundwater quality or availability is not addressed primarily because of the lack of data. We strongly recommend further study of the hydraulic and groundwater quality effects of dewatering. This aspect of mining is coming under increased scrutiny by regulatory agencies at the State and Federal level because of the influence on water quality and availability.

In summary, our approach defines the quality and volume of mine water discharge; outlines hydrographic basins; and calculates flood flows for various return periods ranging from 2 to 25 years in the sub-basin. These flows are then diluted into the average annual flow in the basin and regional basin. The principal objective is to develop a rough estimate of contaminant loads resulting from mine discharge.

3.4.3.1.2 Description of Area

The Grants Mineral Belt of northwestern New Mexico is in the Navajo and Datil sections of the Colorado Plateau physiographic province (Fe31). Characteristic landforms in the study area include rugged mountains, broad, flat valleys, mesas, cuestas, rock terraces, steep escarpments, canyons, lava flows, volcanic cones, buttes, and arroyos (Ki67; Co68). Elevations in the area range from 1,980 m at Grants to an average of 2,160 m near Ambrosia Lake. Just north of Grants is Mount Taylor, the highest point in the region. It rises from Mesa Chivato to an elevation of 3,471 m (Co68).

The study area has a mild, semiarid, continental climate. Precipitation averages 25.4 cm/year, and there is abundant sunshine, low relative humidity, and a comparatively large annual and diurnal temperature range. Average annual precipitation at Gallup, Bluewater, and Laguna is 27.12, 24.55, and 22.31 cm, respectively. In the higher elevations, the average is 51 cm or more because of thunderstorms in July, August, and September and snow accumulations in the winter months (Co68, Go61, Jo63). Only thunderstorms are significant in the lowlands. Heavy summer thunderstorms (40 to 70 in number) of high intensity and local extent can result in 5 cm of rain with local, damaging flash floods.

The watersheds of the Rio San Jose and Rio Puerco encompass 19,037 km^2. Most of the larger communities in the basin are located in the floodplain of the Rio Grande and principal tributaries. Extensive irrigation with surface water occurs in the watersheds of the Rio San Jose, Rio Puerco, and Rio Grande. In the sub-basin, there was no perennial flow before mining and, thus no irrigation, but increasing use is being made of the mine discharge, which is regarded as an asset in a water-short area. Subsequent sections summarize the surface water quantity at some of the principal gauging stations in the Middle Rio Grande Basin and the irrigated areas below these stations. Groundwater is used for essentially all public water supplies as the temperature, quality, and year-round availability are assured. Numerous wells scattered across the landscape, particularly in the stream valleys, are used for stock water and, to a lesser extent, for potable use on the scattered ranches and Indian settlements.

Under completely natural conditions, streams in the study area were distinctly ephemeral, and many of the smaller ones did not experience flow for periods of several years. The Rio Grande experiences peak flows in the April-June period when snowmelt and precipitation cause gradual rises to moderate discharge levels involving large volumes of flow and long durations. Peak discharge rates (volume per time) occur in the summer flash floods.

Construction of dams and conveyance channels to eliminate flooding problems has been extensive. In the tributaries such as the Rio San Jose and upper reaches of the Rio Puerco, there is considerable streamflow regulation to minimize flood damage and maximize use of available water for irrigation.

Conditions in the Ambrosia Lake district with respect to the type of mining operations and discharge of effluent to ephemeral streams are duplicated elsewhere in the Grants Mineral Belt. In the Churchrock district, two mines discharge to the Rio Puerco at rates of 4.7 to 15 m^3/min. Most of the 4.7 m^3/min discharge from one mine is now used in a nearby mill. At Mariano Lake, located between Ambrosia Lake and Churchrock, and at the Marquez and Rio Puerco mines east of Ambrosia Lake, mines are expected to discharge 0.8 to 4.5 m^3/min to various ephemeral streams. Another large mine will soon discharge up to 5.3 m^3/min northward into the San Juan River Basin. In the mid 1980's, construction is expected to begin on five large underground mining projects that will have a combined discharge on the order of 71 m^3/min. Most discharge will be into the San Juan Basin, reflecting the trend of mines becoming deeper and requiring more dewatering as the mining center moves from the south flank of the San Juan Basin into more interior portions.

3.4.3.1.3 Estimate of Sub-basin Flood Flow

Since we use a dilution-model, emphasis is on flow volume rather than peak discharge rate in the sub-basin, basin, and regional basin hydrographic units. Gaging records from the U.S. Geological Survey WATSTORE system (L. Beal, written communication, 1979) provide average discharge rates for runoff events with various return periods and durations. The latter specify the time, in days, and the associated flow rate that will be equaled or exceeded. Flows for arbitrary periods of time ranging from 1 to 183 days are specified. Probability can be stated in terms of N-year recurrence interval. By combining discharge rate (volume per time) and time (partial duration), flow volume can be calculated.

In the ungaged sub-basin, runoff volumes associated with events having return periods of 2, 5, 10, and 25 years were calculated from regression equations developed by the USGS (Bo70). The equations were generated from multiple regression of discharge records from gaged basins against various basin characteristics. These are area (A), precipitation (P_a), longitude at the center of the sub-basin (Lo), soils infiltration index (Si), and mean basin elevation (E_m). Through use of appropriate constants and coefficients (Bo70), flow volumes can be calculated for 1-day and 7-day events with return periods of 2, 5, 10, and 25 years. For the sub-basin, the basic equation has the following form:

$$FV = a\ A^{b_2}\ P_a^{b_9}\ Lo^{b_{14}}\ S_i^{b_{15}}\ E_m^{b_{16}} \qquad (3.10)$$

where $A = 95\ mi^2$

P_a = 2.9 inches

Lo = 7.85 (longitude in decimal degrees minus 100)

S_i = 8.5

E_m = 7.0 thousand feet

Table 3.41 contains the regression coefficients and total flow volume data. Short-term, 1-day and 7-day, events were of main interest because these would be expected to provide greater flushing of contaminants stored at or near the water-substrate interface in the streams receiving mine discharge.

The extent to which mine discharge transforms existing ephemeral streams into perennial ones is evaluated with a crude seepage and evaporation model (see Appendix H). The basic equations and approach are patterned after a similar analysis in the Generic Environmental Impact Statement on Uranium Milling (NRC79b).

Figure 3.16 shows the relationship of the sub-basin, basin, and regional basin boundaries and the principal drainage courses and gaging stations. The confluence of the Rio Puerco and Rio San Jose is shown approximately 55 km closer to the Rio Grande than is actually the case in order to simplify flow routing and to reduce the number of dilution calculations. Table 3.42 summarizes the key characteristics of these basins in terms of catchment area, discharge, and irrigated farmlands downstream from points where mine discharge might be tributary to the streams. Mine discharge occurs in the sub-basin which in turn discharges to the Rio San Jose and then to the Rio Puerco. No mine discharge and no significant runoff are associated with that portion of the basin tributary to San Mateo Creek between the Rio San Jose and the sub-basin. For modeling, flooding within and runoff from the sub-

Table 3.41 Total flow volume for sub-basin floods of 1- and 7-day durations and return periods of 2, 5, 10, and 25 years

Flood Volume	Regression Coefficients						Volume (m^3)
	a	b_1	b_9	b_{14}	b_{15}	b_4	
FV 1,2[(a)]	1.08×10^{-4}	0.931	1.83	-1.43	4.09	-	2.16×10^{4}
FV 1,5	1.27×10^{-3}	0.941	1.40	-1.89	4.07	-	6.23×10^{4}
FV 1,10	5.07×10^{-3}	0.953	1.17	-2.18	4.02	-	1.02×10^{5}
FV 1,25	2.39×10^{-2}	0.972	0.929	-2.51	3.95	-	1.76×10^{5}
FV 7,2	8.60×10^{-7}	0.965	2.36	-1.61	4.22	1.50	5.95×10^{3}
FV 7,5	2.99×10^{-4}	0.904	2.55	-2.09	3.53	-	8.79×10^{3}
FV 7,10	8.97×10^{-4}	0.910	2.37	-2.39	3.61	-	1.43×10^{4}
FV 7,25	3.06×10^{-3}	0.922	2.17	-2.76	3.68	-	2.26×10^{4}

(a) FV 1,2 indicates a flood of 1-day duration and a return period of 2 years.

Table 3.42 Summary of area, discharge, and irrigated acreage for the sub-basin, basin, and regional basin hydrographic units in New Mexico

	USGS Station Number	Area (km^2)	Period of Record Yrs.	Number of km^2 Under Irrigation Below Station	Average m^3/min Discharge (for Period of Record)	Average Annual Discharge (m^3) for Period of Record
Sub-basin	---	246	---	---	---	---
Basins						
Rio San Jose near Grants	3435	5957 (2927 non-contributing)	42	2.43+	11.09	5.83×10^6
Rio Puerco near Bernardo	3530	19037 (2927+ non-contributing)	38	---	81.05	4.26×10^7
Regional Basin						
Rio Grande at Bernardo	3320	49810 (7610 non-contributing)	41	N/C[a]	1649.35	86.69×10^7

---

[a] N/C = Not Calculated.

basin is, in effect, routed without change in flow and quality and allowed to enter the Rio San Jose. Flow from the San Jose is further diluted in the Rio Puerco, then diluted again in the Rio Grande. In actuality, flow from the Ambrosia Lake district rarely, if ever, enters the Rio San Jose because flood volumes are small and infiltration losses are large. This departure from true conditions is justified within the context of the modeling approach used. Basically, the model draws from a specific area but does not attempt to closely duplicate its conditions. If a specific area were exactly represented, the model would still be incorrect to varying degrees for other areas, and the generic value of the assessment would depreciate.

Of special interest is the effect of contaminated flows on irrigation projects present on the Rio San Jose and Rio Grande. An extensive system of dams and conveyance channels regulates flow in the Rio Grande, and partial duration flow data are unavailable. Instead, the average annual flow volume is used to provide the final dilution estimate. For the sub-basin in which the mines are located, flood volumes are calculated using the USGS regression equations (Bo70). The maximum return period for which flows are calculated is 50 years. The remainder of this section first considers the flow or hydraulic aspects of the surface water pathway. Finally, several factors concerning the quality of runoff water are mentioned to balance conservatism and realism in the pathway analysis and, subsequently, in the health effects modeling to follow. The emphasis here is on surface water impacts, and we assume maximum transport for this pathway. The influence of infiltrating mine water is discussed in Section 3.4.3.2.

All of the streams, except the Rio Grande and certain reaches of the Rio San Jose are distinctly ephemeral under natural conditions. In the sub-basin, there is perennial flow because of mine discharge. In Fig. 3.17 are the average monthly and annual discharges for the Rio San Jose and the Rio Puerco in comparison to cumulative annual flow from 14 mines, each discharging at 2 m^3/min. The monthly data reveal pronounced seasonal variations approaching 1 to 2 orders of magnitude. The streams do not show the same seasonal variations, further attesting to varied patterns of runoff, irrigation diversion, and control features such as impoundments and conveyance/irrigation channels. Figure 3.18 shows the percentage of each month during

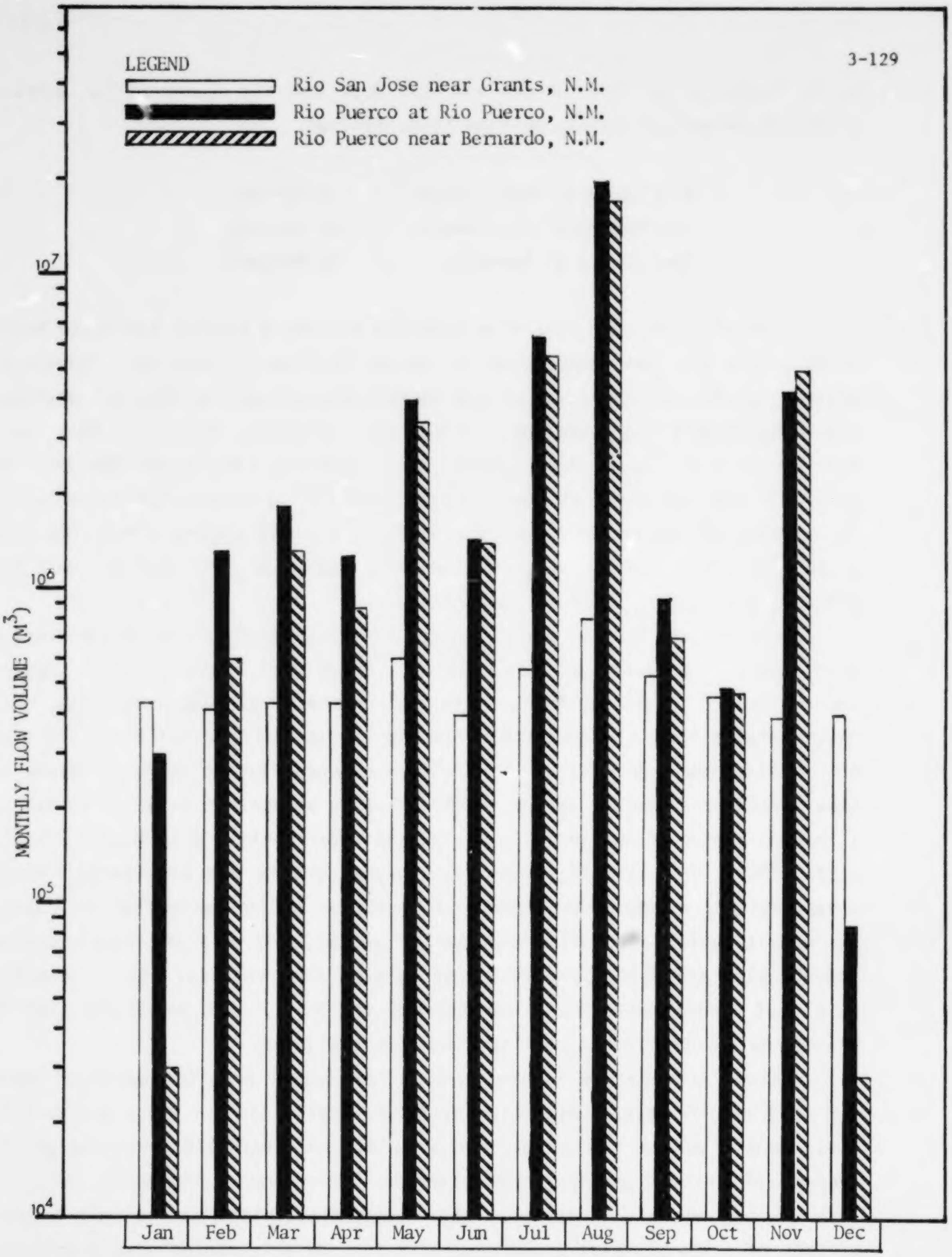


Figure 3.17 Average monthly flows for the period of record for the Rio San Jose and the Rio Puerco in New Mexico(Summarized from flow records provided by L. Beal, U.S. Geological Surve erque).

which there is no flow in the Rio San Jose and Rio Puerco. The average period of annual or monthly no flow is as follows:

Rio San Jose near Grants:	0 Percent
Rio Puerco at Rio Puerco:	45 Percent
Rio Puerco at Bernardo :	71 Percent

It is also assumed that flow from the sub-basin reaches the first major stream, the Rio San Jose, with no change in flow or quality. Runoff is minimal in the lower reaches of San Mateo Creek because of internal drainage and considerable infiltration. Historical evidence indicates that only rarely, if ever, would flood runoff from Ambrosia Lake enter the Rio San Jose. In the interests of conservatism, total flow laden with contaminants is transported to the Rio San Jose. Dilution first occurs within the sub-basin and then, successively, in the Rio San Jose, Rio Puerco, and Rio Grande. The latter is the regional basin.

There is an infinite number of combinations of flood volumes and dilution volumes for the sub-basin, basin, and regional basin streams. Use of average annual discharge volumes in the receiving streams simplifies what would otherwise be a burdensome, confusing series of calculations. Flushing action from the sub-basin is handled on a probabilistic basis in terms of flow duration and return period. Concentration values are based on 14 mines, a loading period of two years, and flow and water quality data shown in Table 3.39. When, for example, 5-year or 10-year events are considered, it is conservatively assumed that events with shorter return periods do not occur. The accretion period remains constant (2 years), and only the return period and duration are varied, resulting in varying flow volumes. It is conceivable that contaminants could concentrate for 3, 4, or 5 years and then be flushed by a 2-year event, but this was not evaluated.

Minimum and maximum return periods for floods from the sub-basin were set at 2 and 25 years, respectively, for several reasons. The 2-year event, i.e., runoff volume over a duration of 1 day or 7 days and occurring on the average of every 2 years, is expected to occur rather frequently over the life of the mines (17 years). The intermediate-sized storms with return periods of 5 or 10 years would result in considerable contaminant transport,

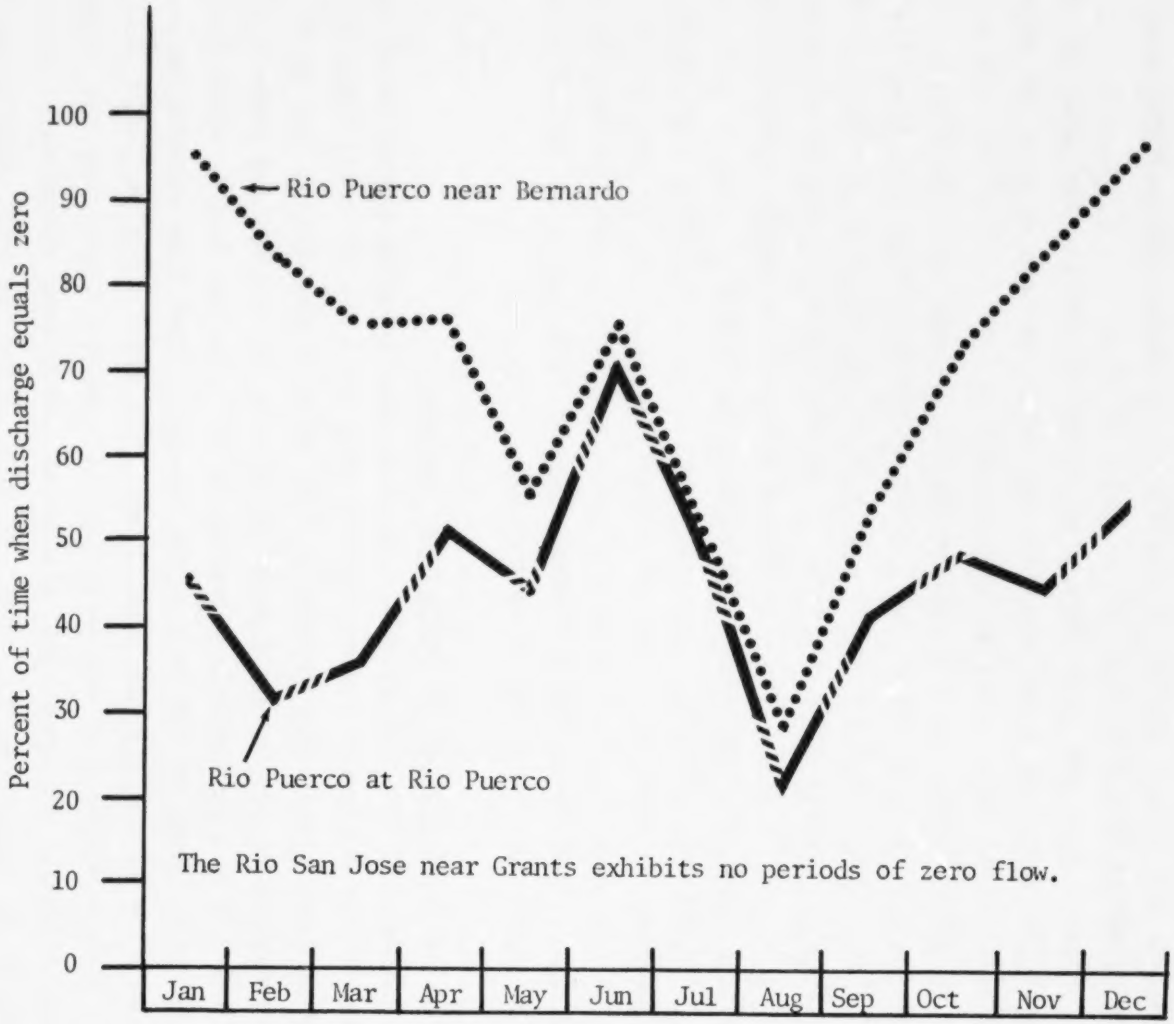


Figure 3.18 Periods of no flow in the Rio San Jose and Rio Puerco (Summarized from flow records provided by L. Beal, U.S. Geological Survey, Albuquerque).

but concentrations would be low owing to dilution and to annual or semiannual scouring provided by smaller floods. The 25-year event is a practical maximum expected to occur during the lifetime of the mining district. Still larger floods, with return periods of 50 or 100 years, can be calculated but are less important because of their infrequent occurrence. Figures 3.19 and 3.20 show calculated flow volumes from the sub-basin for 1-day and 7-day durations and return periods of 2 years to 50 years. The extreme range in flow volume is from 2.16×10^4 m^3 to 2.55×10^5 m^3.

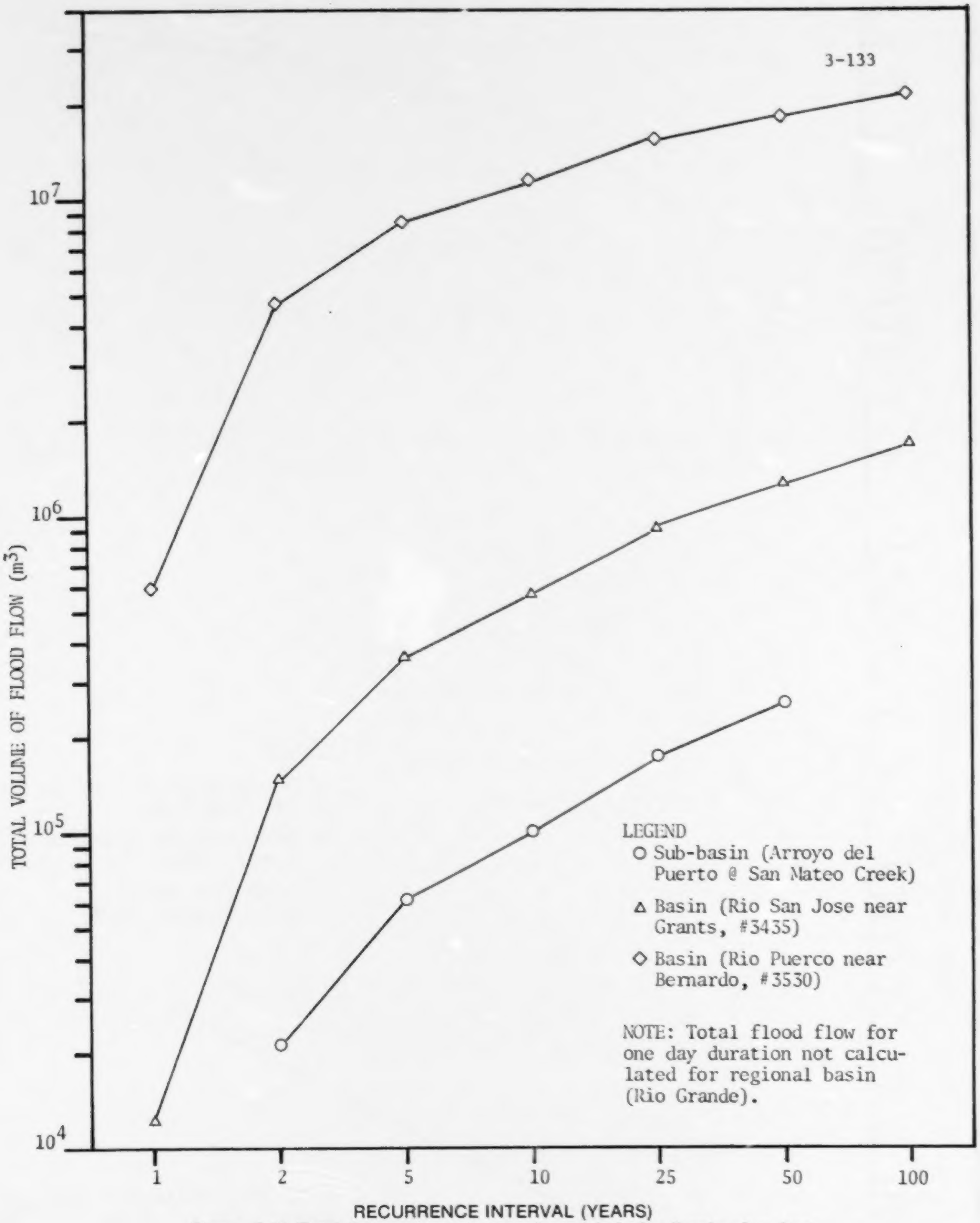
Figures 3.19 and 3.20 show flow values in the Rio San Jose and Rio Puerco for 1-day and 7-day durations and return periods of 1 to 100 years. For the Rio San Jose, 1-day volumes range from 1.24×10^4 m^3 to 1.68×10^6 m^3. The mean annual discharge rate in the Rio San Jose is 11.09 m^3/min. Flow from the Rio San Jose enters the Rio Puerco where corresponding flows (1-day duration) range from 0.6×10^6 to 2.15×10^7 m^3 at the point of inflow to the Rio Grande. Average daily discharge in the Rio Grande seasonally ranges from 8.87×10^5 m^3 to 59.5×10^5 m^3. Average annual flows rather than peak 1-day or 7-day flows were used in the subsequent calculations.

The maximum probability for peak runoff from the sub-basin and resulting contaminant transport is in the summer months, at which time the Rio Puerco has no flow about 22 to 75 percent of the time. Flow in the Rio San Jose and Rio Puerco from June through September ranges from 3.96×10^5 to 1.97×10^7 m^3 per month for the period of record (Fig. 3.17).

3.4.3.1.4 Prediction of Sub-basin Water Quality

Table 3.43 outlines dilutions based on the foregoing discussion of flow patterns and discharges and considering only the 1-day sub-basin flood event with a 2-year recurrence interval. The dilution constant is the ratio of concentration in the receiving water to that in the contaminated (relatively) inflow. It is more commonly expressed as the dilution factor, which is the reciprocal. Thus, in the case of the sub-basin flood flow entering the mean annual flow of the Rio San Jose, there is a 271:1 dilution (Table 3.43).

With development of the foregoing (mine water) source term and surface water pathway, the remaining discussion emphasizes contaminant concentrations in surface water. This, in turn, serves as input data to health effects modeling for the water pathway. Chemical concentrations in the Rio



NOTE: Total flood flow for one day duration not calculated for regional basin (Rio Grande).

Figure 3.19 Total flow volumes in one-day periods for floods of various recurrence intervals in the sub-basin and basins in New Mexico (Summarized from flow records provided by L. Beal, U.S. Geological Survey, Albuquerque).

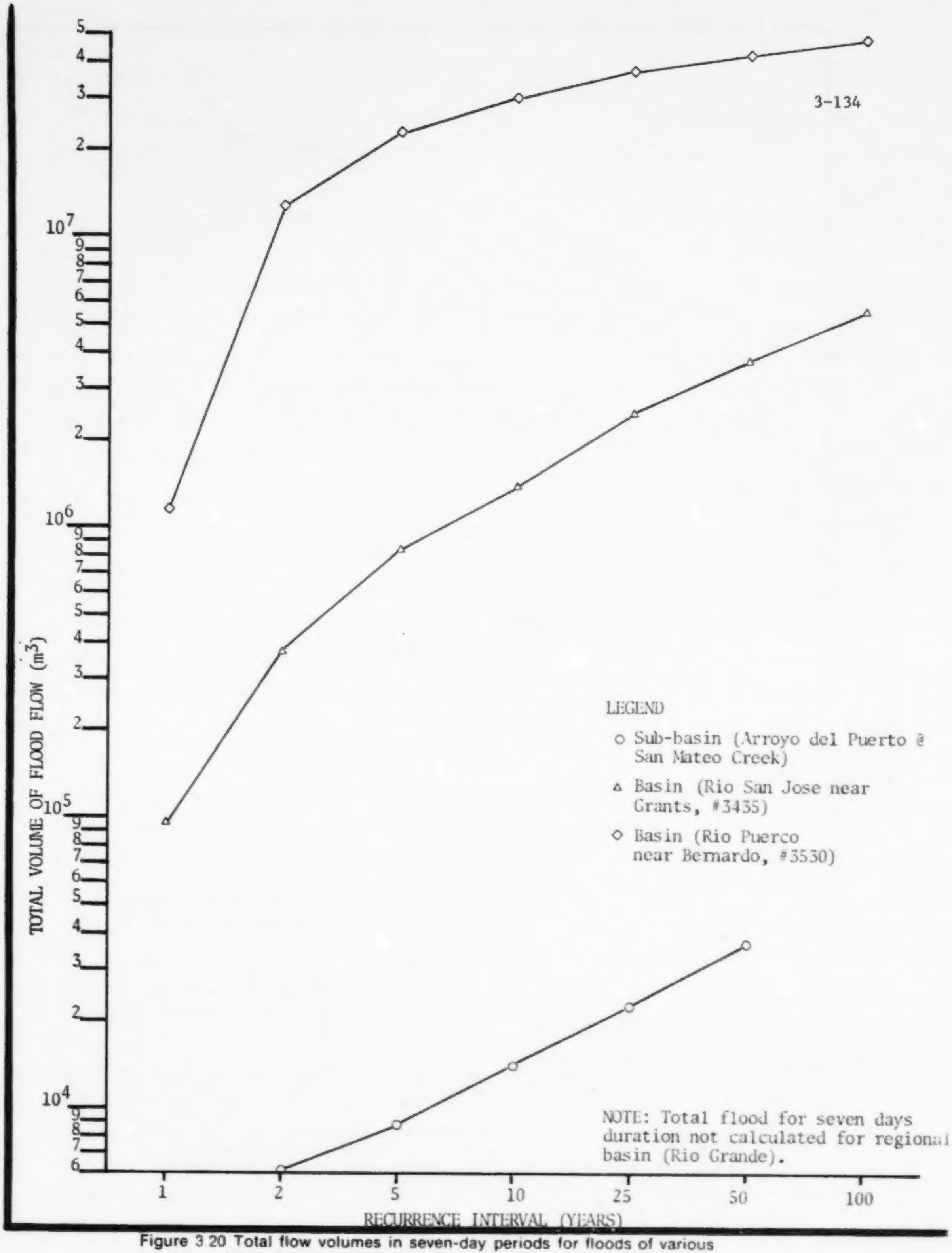


Figure 3.20 Total flow volumes in seven-day periods for floods of various recurrence intervals in the sub-basin and basins in New Mexico (Summarized

Table 3.43 Dilution factors for the Rio San Jose, Rio Puerco, and Rio Grande for 1-day flood flows with a 2-year recurrence interval

Hydrographic Basins	Flow Ratio (m^3/m^3)		Dilution Constant	Dilution Factor
Rio San Jose near Grants[(a)]	$\frac{2.16 \times 10^4}{5.83 \times 10^6 + 2.16 \times 10^4}$	=	0.0037	271
Rio Puerco[(b)]	$\frac{2.16 \times 10^4}{4.26 \times 10^7 + 2.16 \times 10^4}$	=	0.00051	1973
Rio Grande near Bernardo[(c)]	$\frac{2.16 \times 10^4}{86.69 \times 10^7 + 2.16 \times 10^4}$	=	0.000025	40135

(a) Calculated using mean annual flow in the Rio San Jose (near Grants, NM) station:

$$\text{Dilution} = \frac{\text{Sub-basin flood flow}}{\text{Rio San Jose flow + Sub-basin flood flow}} .$$

(b) Assumes Rio San Jose enters the Rio Puerco at Bernardo:

$$\text{Dilution} = \frac{\text{Sub-basin flood flow}}{\text{Rio Puerco flow (includes Rio San Jose flow) + Sub-basin flood flow}} .$$

(c) $$\text{Dilution} = \frac{\text{Sub-basin flood flow}}{\text{Rio Puerco flow + Rio Grande flow (at Bernardo) + Sub-basin flood flow}} .$$

San Jose, Rio Puerco (at Bernardo), and in the Rio Grande (near Bernardo) are shown in Table 3.44 along with 1-day and 7-day flood flow volumes from the sub-basin for return periods of 2, 5, 10, and 25 years. These flood volumes are diluted into the mean annual flow of the Rio San Jose (near Grants), Rio Puerco (at Bernardo), and Rio Grande (near Bernardo). The principal reason for using mean annual flow is that the radiation dose and health effects model (Section 6.0) stresses estimating average annual dose to the population over the duration of mining activity.

For example, the 1-day duration flood flow (with a 2-year return period) contains 1920 mg/ℓ uranium, which decreases to 7.09 mg/ℓ in the Rio San Jose and 0.973 mg/ℓ in the Rio Puerco. Because of the short duration of most floods in the sub-basin, there is little difference in flow volume and, thus, dilution between the 1-day and 7-day events. With progressive dilution downstream, the difference in size between sub-basin floods of varying durations and return periods becomes insignificant relative to the mean annual flow volumes of the basin and regional basin streams. As a result, concentrations tend to reach a minimum and remain unchanged at this degree of accuracy.

As in the case of the Wyoming surface-mine scenario, we assume that most contaminants in the mine water collect on or near the land surface and are available for transport. This assumption is open to question, but field data are scarce to support contentions as to the fraction of contaminant load that becomes unavailable. For example, extensive field studies along the Animas, San Miguel, and Dolores Rivers in Colorado concluded that "...once radium becomes a part of a stream's environment, it constitutes a relatively long-term and continuous source of water and aquatic biota contamination" (Si66). However, cessation of uranium mill discharges to the Colorado River tributaries effectively negated this source, which is now believed to be buried behind the Lake Powell and Lake Mead impoundments. Similarly, dissolved radium reverts to background levels of several picocuries per liter in natural streams receiving mine water in Colorado and New Mexico. Although it is likely that flood waters resuspend precipitates and sediments with sorbed radium, laboratory experiments (Sh64; Ha68) indicate that only minor re-solution takes place. This phenomenon is supported by recent surface water data collected in the Grants Mineral Belt of New Mexico (Ku79). Therefore, concentrations of dissolved radium in flood water are

Table 3.44 Annual contaminant loading from 14 uranium mines and resulting concentrations in sub-basin floods and in the average annual flow of the Rio San Jose, Rio Puerco, and Rio Grande

Contaminant concentration in mine effluent (mg/ℓ except as noted)	Mass available for transport (kg/yr except as noted)(a)	1- and 7-day flood flow volumes (m^3) and contaminant concentrations associated with return periods of 2 to 25 years(b)							
		1-Day				7-Day			
		$V_2=2.16\times10^4$ C_2	$V_5=6.23\times10^4$ C_5	$V_{10}=1.02\times10^5$ C_{10}	$V_{25}=1.76\times10^5$ C_{25}	$V_2=5947$ C_2	$V_5=8794$ C_5	$V_{10}=1.43\times10^4$ C_{10}	$V_{25}=2.26\times10^4$ C_{25}
Total Uranium 1.41	1480	1920 7.09 0.973 0.0456	665 7.03 0.971 0.0456	406 6.98 0.970 0.0455	235 6.89 0.967 0.0455	6970 7.10 0.973 0.0456	4710 7.09 0.972 0.0455	2900 7.10 0.973 0.0456	1830 7.07 0.970 0.0456
Radium-226 13.7 pCi/ℓ	0.00144 Ci/yr	1870 6.9 0.95 0.044	647 6.8 0.95 0.044	395 6.8 0.94 0.044	229 6.7 0.94 0.044	6780 6.9 0.95 0.044	4580 6.9 0.95 0.044	2820 6.9 0.95 0.044	1780 6.9 0.94 0.044
Lead-210 14.6 pCi/ℓ	0.00153 Ci/yr	19800 73.1 10.0 0.470	6880 72.7 10.0 0.471	4200 72.2 10.0 0.471	2430 71.2 10.0 0.470	72000 73.4 10.0 0.471	48700 73.3 10.1 0.471	30000 73.4 10.1 0.472	19000 73.4 10.1 0.472
Cadmium 0.007	7	9 0.03 0.004 0.0002	3 0.03 0.004 0.0002	2 0.03 0.005 0.0002	1 0.03 0.004 0.0002	30 0.03 0.004 0.0002	20 0.03 0.004 0.0002	10 0.02 0.003 0.0002	9 0.03 0.005 0.0002
Arsenic 0.012	13	17 0.063 0.0086 0.00040	5.8 0.061 0.0085 0.00040	3.6 0.062 0.0086 0.00040	2.1 0.062 0.0086 0.00041	61 0.062 0.0085 0.00040	41 0.062 0.0085 0.00040	25 0.061 0.0084 0.00039	16 0.062 0.0085 0.00040
Selenium 0.076	80	100 0.38 0.053 0.0026	36 0.38 0.052 0.0026	22 0.38 0.052 0.0026	13 0.37 0.052 0.0026	376 0.38 0.053 0.0026	254 0.38 0.053 0.0026	156 0.38 0.053 0.0026	99 0.38 0.053 0.0026

Table 3.44 (continued)

Contaminant concentration in mine effluent (mg/ℓ except as noted)	Mass available for transport (kg/yr except as noted)[a]	1- and 7-day flood flow volumes (m^3) and contaminant concentrations associated with return periods of 2 to 25 years[b]							
		1-Day				7-Day			
		V_2=2.16x10^4 C_2	V_5=6.23x10^4 C_5	V_{10}=1.02x10^5 C_{10}	V_{25}=1.76x10^5 C_{25}	V_2=5947 C_2	V_5=8794 C_5	V_{10}=1.43x10^4 C_{10}	V_{25}=2.26x10^4 C_{25}
Molybdenum	300	390	130	82	48	1400	960	590	370
0.29		1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4
		0.20	0.19	0.20	0.20	0.20	0.20	0.20	0.20
		0.0093	0.0089	0.0092	0.0091	0.0092	0.0093	0.0091	0.0092
Barium	850	1100	380	230	140	4000	2700	1700	1100
0.81		4.1	4.0	4.0	4.1	4.1	4.1	4.2	4.2
		0.56	0.55	0.55	0.58	0.56	0.56	0.57	0.58
		0.026	0.026	0.026	0.027	0.026	0.026	0.027	0.027
Zinc	45	58	20	12	7.2	210	140	88	56
0.043		0.21	0.21	0.21	0.21	0.21	0.21	0.22	0.22
		0.029	0.029	0.029	0.030	0.029	0.029	0.030	0.030
		0.0014	0.0014	0.0014	0.0014	0.0014	0.0014	0.0014	0.0014
Sulfate	1.22 x 10^5	1.58 x 10^5	5.48 x 10^4	3.35 x 10^4	1.94 x 10^4	5.74 x 10^5	3.88 x 10^5	2.38 x 10^5	1.51 x 10^5
580		584	580	574	568	586	584	582	584
		80	80	80	80	80	80	80	80
		3.8	3.8	3.7	3.8	3.8	3.8	3.7	3.8
Total Suspended Solids	29000	38000	13000	8000	4600	140000	92000	57000	36000
27.8		140	140	140	130	140	140	140	140
		19	19	19	19	20	19	19	19
		0.90	0.89	0.90	0.89	0.92	0.89	0.90	0.89

[a] Mass values shown are on an annual, per-mine basis.

[b] V_r and C_r refer respectively to flood volume, in cubic meters, and concentration in runoff for an r-year flood. Concentrations are in mg/ℓ, except for radium-226 and lead-210, which are in pCi/ℓ. Concentrations shown are from accretion or loading in the sub-basin for 2, 5, 10, 25 years, yielding the first value shown in each set. The next three values below this initial value represent, in downward order, concentrations in the flood flow as diluted by the mean annual flow in 1) the Rio San Jose near Grants (5.83 x 10^6 m^3), 2) the Rio Puerco at Bernardo (4.26 x 10^7 m^3), and 3) the Rio Grande near Bernardo (86.69 x 10^7 m^3).

Note.--Assumptions: Mines discharge continuously at a rate of 2.0 m^3/min. Concentrations are the average of those shown in Table 3.39. Except for radium and sulfate, all suspended and dissolved contaminants remain in or on the stream sediments and are mobilized by flood flow. Twenty percent of the sulfate and 10 percent of the radium are available for resolution.

arbitrarily set at 0.00144 Ci/yr or 10 percent of the annual loading from the model mine.

Sulfate is also considered an important exception in the total "transport" concept. Because sulfate can be a highly mobile anion, it is assumed that 80 percent of the load enters the shallow groundwater reservoirs and 20 percent is available for solubilization and chemical transport in surface flows. No distinct pattern of groundwater contamination from mine water, per se, was documented in an earlier Grants Mineral Belt survey (EPA75), but recent data from the State indicate groundwater deterioration as a result of mine drainage (J. Dudley, New Mexico Environmental Improvement Division, oral communication, 1979). It is likely that considerable fractionation of other stable and radioactive trace elements occurs, but field data specific to the uranium mining regions are quite scarce, with the exception of Texas (He79), where only stable elements were studied. Because of our imperfect, non-predictive understanding of trace element transport in aqueous systems, our analysis assumes total transport for most constituents in lieu of numerous, equally unfounded assumptions for resuspension factors, fractionation, etc. Floods of 1-day and 7-day duration and return periods of 2, 5, 10, and 25 years are arbitrarily selected as providing the necessary flushing action associated with intense, short-term runoff events. It is likely that storms of shorter (less than 1-day) duration and possibly greater discharge rate also transport contaminants. The flow volume and thus the dilution cannot be estimated for these events.

Calculated water quality in basin and regional basin streams is shown in Table 3.45 along with established and suggested standards for selected contaminants. For uranium, concentrations in the basin exceed the suggested limits based on chemical toxicity and radiotoxicity. Radium-226/228 exceeds the standard in the basin but is well below the standard for the regional basin. The same is true for sulfate, cadmium, arsenic, barium, and selenium. Zinc is the only contaminant consistently below the potable and irrigation water standards. As in the case of the surface mine scenario for Wyoming, uranium is apparently well above suggested limits and warrants further study, as do the stable toxic elements in the basin area(s) closest to the mining centers.

With the exception of radium-226 and sulfate, the concentrations of radionuclides and other parameters shown in Tables 3.44 and 3.45 reflect no

Table 3.45 Comparison of potable and irrigation water standards and surface water quality affected by underground mine drainage

Parameter	Range of contaminant concentrations in flood flow affected by mine discharge(a) Basin Min.	Basin Max.	Regional Basin Min.	Regional Basin Max.	Potable water standards (mg/ℓ)(b) Maximum Permissable Concentration	Recommended Limiting Concentration	Irrigation(c) Recommendations for maximum concentration for continuous use on all soils (mg/ℓ)
Total U	6.9	7.1	0.045	0.046	0.015/3.5/0.2(d)	----	---
Ra-226 + 228	6.7	6.9	0.044	0.044	----	5 pCi/ℓ	5 pCi/ℓ
TSS	130	140	0.89	0.92	---	---	---
Sulfate	574	584	3.7	3.8	---	250	200
Zn	0.21	0.22	0.0014	0.0014	---	5.0	2.0
Cd	0.03	0.03	0.0002	0.0002	0.01	---	0.010
As	0.061	0.063	0.00039	0.00041	0.05	0.01	0.10
Ba	4.0	4.2	0.026	0.027	1	---	---
Se	0.37	0.38	0.0026	0.0026	0.01	---	0.02

(a) Concentrations in milligrams per liter, except Ra-226 -228 which are in picocuries per liter. Data shown apply to the Basin (Rio San Jose near Grants) and Regional Basin (Rio Grande near Bernardo) streams (Table 3.44).

(b) Sources: U.S. Environmental Protection Agency (EPA76) and, in the case of uranium, suggested guidance from the National Academy of Sciences (NAS79) to the USEPA and from USEPA, (Office of Drinking Water) to the State of Colorado (La79).

(c) Source: (NAS72).

(d) 0.015 mg/ℓ :Suggested maximum daily limit based on radiotoxicity for potable water consumed at a rate of 2 liters per day on a continuous basis
3.5 mg/ℓ: Suggested maximum daily limit based on chemical toxicity and intake of 2 liters in any one day
0.21 mg/ℓ: Suggested maximum daily limit based on chemical toxicity and intake of 2 liters per day for 7 days

reductions for ion exchange, precipitation, or sorption. Rather, a simple dilution model is used in which the mass loading from mine discharge is calculated as the product of concentration and discharge (volume). There are problems with this approach. In some cases, the calculated concentrations in flood waters probably exceed the solubility limits, as in the case of sulfate in the presence of barium. In other instances, precipitation of barium sulfate or iron and manganese hydroxides might greatly reduce the concentration of radium and uranium, both of which would coprecipitate. Thus the stream concentrations shown in Table 3.44 are probably high (conservative). To improve the analysis, additional comparisons or parallels were drawn using mill tailings solutions and stream water quality as affected by mine drainage and a mill tailings spill.

Contaminant concentrations in uranium mill tailings liquids provide an upper limit estimate of runoff concentrations insofar as the solvent action of tailings solutions maximize dissolution of minerals present in the ore (J. Kunkler, USGS, written communication, 1979). Table 3.46 is a compilation of mill tailings water quality data from numerous previous reports and summarized by EPA and USGS staff (Ka79; Ku79). It is apparent that there are wide variations as a function of mining region and whether an acid or alkaline leach mill circuit is used. The Nuclear Regulatory Commission (NRC79b) assumption for the composition of a "typical" acid leach mill is shown along with other average or representative analyses. A conservative (worst quality) analysis for uranium mill pond water quality is estimated as follows (Table 3.47) and compared to the average concentrations calculated from the mixing of mine effluent and flood volumes (Table 3.44).

The data in Table 3.47 suggest that calculated concentrations in the sub-basin almost without exception exceed those in uranium mill tailings solutions. Thus, the calculated values are probably erroneously high. Calculated concentrations in flood waters of the basin and regional basin streams are considerably less and are in rough agreement with field data, at least for the stable constituents. Radium-226 and lead-210, however, still seem excessively high considering the various natural processes of sorption, precipitation, and so on. To understand the degree to which natural streams transport contaminants, we reviewed water quality data from selected New Mexico streams receiving mine drainage.

Table 3.46 Radiochemical & stable element/compound water quality for selected acid & alkaline leach uranium mill tailings ponds in the United States

Tailings Pile Location	U	Th-230	Ra-226	Pb-210	Po-210	As	Mn	Cu	Se	Mo	V	SO	Na	Fe	TDS	NH	Ca	NO	Cl
	(mg/ℓ)		(pCi/ℓ)								(mg/ℓ)								
1. Split Rock, WY (acid)	10.5	41600	4800	---	940	1.1	15.5		0.2	1	0.05			280	11810	374	560	43.5	65
2. Canon City, CO (acid)	---	---	---	---	---	10.1	25.0	18	0.6	190	7.1	34000	19000	280	77400	---	380	140	6500
3. Moab, UT	2.0	50	100	---	---	7.0	---	--	---	---	---	100	---	---	150000	---	---	---	300
4. United Nuclear, NM (acid)(a)	14	---	38	---	---	---	50	3	0.005	3	30	---	300	1000	---	---	700	---	---
5. Anaconda Inj. Well Feed, NM	130	---	53	---	---	---	340	--	0.03	---	6.3	4900	1200	---	---	69	---	7.4	---
6. Kerr-McGee, NM (acid)(a)	32	---	58	---	---	---	30	5	0.18	7	10	---	500	1000	---	---	300	---	---
7. UN-HP, Grants, NM (alkaline)	150	---	52	---	---	---	---	--	0.92	70	6.8	4300	4300	---	---	4.4	---	4.4	2
8. Humeca, WY (acid)	68.4	110	240	---	---	---	0.4	0.1	---	---	---	6500	11700	0.5	---	460	---	16	16000
9. USNRC-Uranium Milling EIS(acid)	8.0	150000	400	400	400	0.2	500	50	20	100	0.1	30000	500	1000	35000	500	500	---	300
10. Representative acid millpond in New Mexico(a)	---	---	---	---	---	---	---	--	---	---	---	44000	---	170	---	300(b)	---	---	1800
"Average" (Exclusive of 9 and 10)	58	13920	760	---	---	6	160	6.5	0.32	54	10	10000	6200	510	80000	227	485	40	1700
Maximum value: "Average" versus NRC GEIS	58	150000	760	400	400	6	500	50	20	100	10	30000	6200	1000	80000	500	500	40	1700

(a) Source: Ku79.
(b) Ammonium ion.

Table 3.47 Summary of flood runoff water quality and uranium millpond quality

Parameters	Concentration in uranium mill tailings solution	Concentration in flood waters of the sub-basin[a]	Concentration in flood waters of the Rio San Jose[a]
Uranium (mg/ℓ)	58	235 - 6970	7
Radium-226 (pCi/ℓ)	760	229 - 6780	6.9
Lead-210 (pCi/ℓ)	400	2430 - 72000	73
Polonium-210 (pCi/ℓ)	400	NC[b]	NC
Arsenic (mg/ℓ)	6	2.1 - 61	0.062
Manganese (mg/ℓ)	500	NC	NC
Copper (mg/ℓ)	50	NC	NC
Selenium (mg/ℓ)	20	11 - 220	0.34
Molybdenum (mg/ℓ)	100	48 - 2400	1.4
Vanadium (mg/ℓ)	10	NC	NC
Sulfate (mg/ℓ)	30,000	$9.7 \times 10^4 - 2.87 \times 10^6$	2901

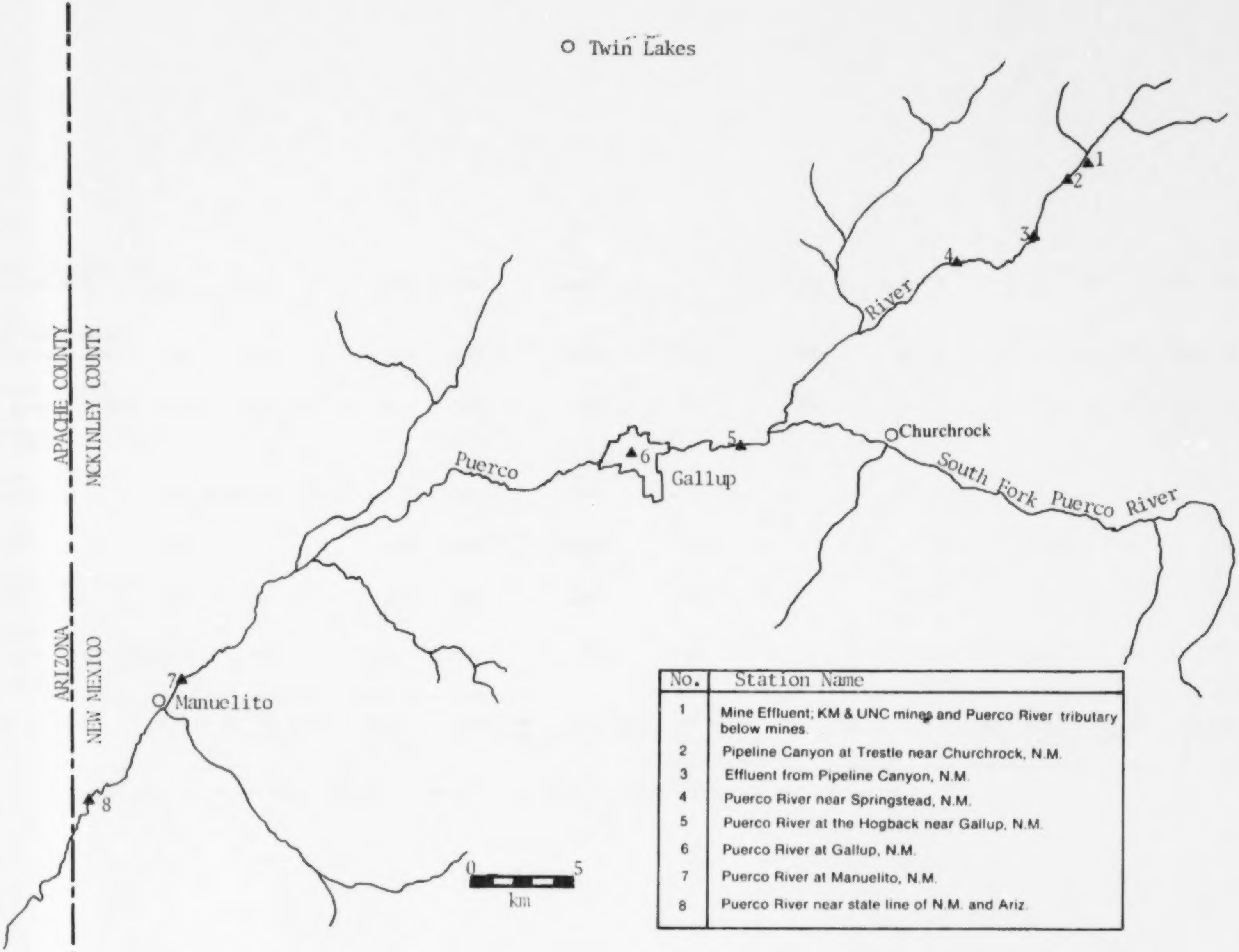
[a] Refer to Table 3.44.

[b] Not calculated.

The USGS, by water sampling in the Churchrock area of New Mexico (J.L. Kunkler, USGS, written communication, 1979), determined water quality in an ephemeral stream receiving rather large and continuous mine discharges. Data are also available from the Schwarzwalder Mine near Golden, Colorado (EPA72). Until 1972, this mine discharged effluent high in uranium, radium, and trace elements to Ralston Creek and subsequently to two lakes/reservoirs used for irrigation and potable supply (Section 3.2.3.2.1).

The way in which surface runoff water quality is created or affected by mine discharge is complex. In the Churchrock area, numerous water quality changes occur as the mine discharges flow toward Gallup (Fig. 3.21 and Table 3.48). As in other uranium mining areas in New Mexico, stream volume constantly decreases with flow distance, but water quality changes are erratic. Infiltration, discussed in more detail in the following section of the report and in Appendix H, amounts to about 90 percent or more of the water loss. The balance is by evaporation. On a percentage basis, similar losses occur in the principal drainage courses in Ambrosia Lake. Dissolved Ra-226 decreases from 30 to 0.88 pCi/ℓ in a reach of 9.2 km and, on a later date, from 14 to 0.95 pCi/ℓ in a distance of 26.7 km. Based on the limited flow and water quality data, it appears that radium is strongly sorbed onto the stream sediments. In October 1975, soluble uranium decreased from 1150 to 740 μg/ℓ in the reach immediately below the mine discharges, yet in July 1977 and May 1978 uranium increased in the downstream direction from 580 to 860 μg/ℓ and from 970 to 2800 μg/ℓ. These changes bear no consistent relation to fluctuations in dissolved or suspended solids along the flow path. Both of the latter parameters appear to increase in the direction of flow and may be a result of flash floods in lower reaches of the basin. Uranium appears to undergo little change and may actually increase in the downstream direction. Of the stable trace elements, vanadium, selenium, iron, molybdenum, and zinc show no consistent change with distance.

A third approach used to assess surface runoff quality involved a brief review of some of the data collected to monitor a July 1979 tailings accident in New Mexico. The mill tailings dam at the Churchrock mill breached and dumped 223,000 m^3 of liquid and 1,000 metric tons of solids into the Rio Puerco drainage system. The catastrophe immediately spurred numerous water quality studies by State and Federal agencies. Numerous inter-



No.	Station Name
1	Mine Effluent; KM & UNC mines and Puerco River tributary below mines.
2	Pipeline Canyon at Trestle near Churchrock, N.M.
3	Effluent from Pipeline Canyon, N.M.
4	Puerco River near Springstead, N.M.
5	Puerco River at the Hogback near Gallup, N.M.
6	Puerco River at Gallup, N.M.
7	Puerco River at Manuelito, N.M.
8	Puerco River near state line of N.M. and Ariz.

Figure 3.21 Principal streams and surface water sampling stations in the Churchrock and Gallup areas

Table 3.48 Flow and water quality in the Puerco River near Churchrock and Gallup, New Mexico

Location and (Station Number)	m^3/min	U nat. µg/ℓ as U_3O_8	Ra-226 pCi/ℓ	Total Solids, mg/ℓ		Suspended solids, metric tons per day	Concentrations µg/ℓ									
				Dissolved	Suspended		Ba	Cd	Cr	Pb	Mo	V	Zn	Se	As	Fe
Oct. 16, 1975																
Puerco River tributary below mines - (1)	14.5	1150	30	430	410	9.5	-	-	-	-	-	21	-	27	-	20
Puerco River near Springstead, NM -(4)	12.4	740	0.88	480	1600	8.67	-	-	-	-	-	13	-	25	-	30
Puerco River at Gallup, NM - (6)	5.11	---	0.52	640	2300	5.14	-	-	-	-	-	5.7	-	26	-	40
Puerco River at Manuelito - (7)	6.8(est)	540	0.25	800	2800	---	-	-	-	-	-	-	-	-	-	-
July 6, 1977																
Puerco River tributary below mines - (1)	11.55	580	14	410	260	---	800	1	0	6	-	-	0	25	1(3)	10
Puerco River at the Hogback, near Gallup, NM - (5)	6.47	860	0.95	520	15000	---	100	1	0	11	-	-	50	20	1(19)	80
Puerco River near State line (NM/AZ) - (8)	15.5	83	0.27	600	44000	---	1700	4	0	2	-	-	30	5	6(7)	90

Table 3.48 (Continued)

Location and (Station Number)	m^3/min	U nat.		Ra-226	Total Solids, mg/ℓ		Suspended solids, metric tons per day	Concentrations µg/ℓ									
		µg/ℓ	as U_3O_8	pCi/ℓ	Dissolved	Suspended		Ba	Cd	Cr	Pb	Mo	V	Zn	Se	As	Fe
May 25, 1978																	
Effluent from Kerr McGee and United Nuclear Mines, Churchrock, NM- (1)	10.9	807		2.6	---	---	---	-	-	-	-	12	19	-	4	-	-
Effluent from Pipeline Canyon, NM - (3)	9	2800		1.5	---	---	---	-	-	-	-	820	28	-	110	-	-
Puerco River near Springstead, NM - (4) (sampled 5/18/78)	10.88	1100		0.8	---	---	---	-	-	-	-	12	16	-	0	-	15
July 11-12, 1978																	
Pipeline Canyon at trestle near Churchrock, NM - (2)	14.45	940		8.6	---	---	---	-	-	-	-	230	11	-	-	-	540
Effluent from Pipeline Canyon, NM - (3)	14.3	1120		1.3	---	---	---	-	-	-	-	260	6	-	11	-	70
Puerco River near Springstead, NM - (4)	---	1130		2.2	---	---	---	-	-	-	-	240	9	-	13	-	40

Source: New Mexico District office of the U.S. Geological Survey (Peter Frenzel, written communication, 1979 and Kunkler, 1979).

pretations of the data have led to some confusion, compounded in some instances by inconsistent sample collection and preservation. However, several general findings seem true. Dissolution of stable and radioactive trace contaminants in flood waters does not seem significant providing that pH of the flood is in the range of 4 to 7. After several days, the mill tailings liquid was diluted and neutralized and contaminant concentrations decreased -- sometimes to levels lower than before the accident (J. Kunkler, USGS, written communication, 1979). At a downstream sampling station near Gallup, some 30 kilometers from the spill, dissolved uranium and radium-226 about 36 hours after the spill were 3.1 mg/ℓ and 0.95 pCi/ℓ, respectively. Suspended sediments contained 19 ppm uranium and 0.72 pCi/g radium-226. For the latter, this is less than background.

The surface water quality data pertaining to discharge of mine effluents and to the July 1979 spill seem to indicate rapid and thorough removal of radium-226 as a result of sorption, precipitation, pH adjustment, etc. However, stream sediment analyses in the Grants Mineral Belt are scarce, and there are no analyses of suspended solids in flood waters. Stream-bed sediment analyses by the USGS indicate less sorbed radium-226 and uranium than expected (Ku79). During this spill incident, uranium and selenium were relatively mobile in surface streams.

From the foregoing review of the literature and field data and preliminary calculations of runoff quality (Table 3.44), the following general conclusions are offered:

1. Radium-226 is removed from surface water in the New Mexico study area at rates of 0.5 to 3 pCi/ℓ per kilometer of stream. Final concentrations are on the order of 0.25 pCi/ℓ. Resolution in successive surface flows occurs, but it is not significant.

2. Uranium and certain stable trace elements, such as selenium, vanadium, molybdenum, and iron, show no consistent reduction with flow distance and may show an increase, at times.

3. Considerable more data collection is needed to understand the fate of dissolved and suspended contaminants from mine drainage. The present data base is rather limited in terms of sampling frequency, variety of contaminants measured, and types of measurements, for example, suspended solids analyses for flood waters.

4. With the exception of radium-226, the preliminary calculations of runoff quality in Table 3.44 are believed to be a first approximation of field conditions. Additional studies specific to the principal mining districts are needed.

5. Dissolved radium-226 concentrations in runoff are believed to be several picocuries per liter or less under natural conditions.

6. Uranium is fairly mobile and probably the most significant radionuclide in uranium mine effluent.

3.4.3.2 Impacts of Seepage on Groundwater

The principal use of groundwater in the immediate area of the mines is for stock water. Wells in the highland areas are typically one to two hundred meters deep and completed in underlying bedrock strata (Co68; Ka75). Contamination of such wells by mine discharge is considered extremely unlikely. Shallow wells are few in number and located along major drainages that are typically ephemeral. Such shallow wells are susceptible to contamination if located downgrade from mine discharges. Municipal water supplies are usually developed from wells because groundwater is consistently available and has acceptable suspended and dissolved mineral contents. The aquifers tapped by municipal wells are mostly either quaternary lava flows or deeper mesozoic sandstone and carbonate sequences. Considering the distance from the mining centers to the communities and the hydrogeologic conditions, it is unlikely that mining will cause measurable deterioration of municipal water quality. The greatest likelihood for contaminated groundwater is in the shallow, alluvial aquifer beneath streams receiving mine drainage. It is extremely unlikely that water quality in deeper, artesian aquifers will be adversely affected by mine discharge or overland flow affected by solid wastes. Shallow wells in these locations have been constructed in the past, but there are only a few and they are used for stock watering. It is possible that recharge of substantial quantities of mine water to the shallow aquifer will encourage additional use of it, in which case water quality will be of concern.

Table 3.49 shows average and extreme concentrations of various common and trace constituents in groundwater and other measures of water quality. The data are composited from a previous study (EPA75) and from unpublished analyses by the New Mexico Environmental Improvement Division (J. Lazarus, NMEID, oral communication, 1979). We have categorized the data according to principal aquifers, which are in areas where the groundwater is not believed to be contaminated by mining. Because it is common for a well to tap more than one aquifer, the differences in water quality in Table 3.49 are approximate at best. The data reveal no sharp differences in water quality amongst the three major aquifers. The San Andres Limestone, a major aquifer for municipal and industrial uses in the Grants and Milan areas, has equal or greater concentrations of most constituents as compared to the Westwater Canyon Member and Gallup Sandstone units, which are closely associated with uranium mineralization.

Theoretical analysis of radionuclide transport in groundwater beneath and adjacent to a uranium mill tailings pond reveals very limited migration of radionuclides in groundwater (Se75). Using a seepage rate of 4×10^{-7} cm/sec and a 10 percent loss of soluble radionuclides, numerical solutions for steady state flow and transport into unconsolidated sand for periods of 5 years and 20 years reveal up to several meters movement of radium-226, thorium-230/ 234, uranium, and lead-210 after 20 years of leaching. For example, radium in groundwater to a depth of 3 meters is 10 percent of that in the tailings pond. Because the other isotopes tend to have even greater sorption, migration distances are further reduced. Although field studies at three uranium mill tailings piles in the Grants Mineral Belt substantiate only local migration of radionuclides (EPA75), extensive lateral migration of stable chemical species has been observed at uranium mills in Colorado, Wyoming, and Washington (Ka79, Ka78a, He79). For example, with respect to the old Cotter uranium mine at Canon City, Colorado, the Colorado Department of Health has stated in its Final Executive Licensing Summary, August 17, 1979, that "contamination attributed to tailings liquid was observed in an off-site water well ten years after the mill began depositing tailings, a [migration] rate of over five hundred feet per year." With respect to the same site, one researcher has stated that "the soluble uranium content of Lincoln Park ground waters is highly elevated with respect to Arkansas River water and exceeds suggested thresholds below which ecological and health effects are not expected. Molybdenum concentrations in these ground waters greatly exceed irrigation standards as well as the ALG based on health and ecological effects...." (Dr79). Near neutral pH and relatively low concen-

Table 3.49 Groundwater quality in principal aquifers in the Grants Mineral Belt, New Mexico

	Aquifer		
Parameter	San Andres Limestone	Westwater Canyon Member, Morrison Fm. and Gallup Sandstone	Quaternary Alluvium, Tertiary Volcanics, and Chinle Formation
pH	7.2(a) (6.9 - 7.5)	7.9 (6.7 - 9.15)	7.6 (6.25 - 8.8)
Spec. cond. µmhos/cm	1900 (720 - 3500)	1800 (550 - 4250)	1715 (700 - 4000)
TDS	1680 (490 - 4500)	1160 (340 - 2300)	1240 (490 - 3800)
Cl mg/ℓ	98 (< 0.2 - 270)	15 (0 - 98)	57 (6.2 - 260)
Se mg/ℓ	0.31 (0.01 - 1.52)	0.02 (0.01 - 0.13)	0.59 (0.02 - 1.06)
V mg/ℓ	0.88 (0.4 - 1.3)	0.3 (0.3 - 0.3)	0.55 (0.3 - 1.3)
Radium-226 pCi/ℓ	0.47 (0.11 - 1.92)	0.71 (0.07 - 3.7)	0.22 (0.05 - 0.72)
Uranium, mg/ℓ	1.31 (0.04 - 2.6)	0.35 (0.02 - 1.0)	4.72 (0.07 - 14)
Th-230, pCi/ℓ	0.12 (0.017 - 0.52)	0.030 (0.015 - 0.053)	0.212 (0.018 - 0.65)
Th-232, pCi/ℓ	0.11 (0.0053-0.54)	0.015 (<0.01-0.036)	0.123 (0.0094-0.99)
Po-210, pCi/ℓ	0.75 (0.070 - 2.3)	0.42 (0.19 - 0.79)	0.193 (0.010 - 0.55)

(a) Mean and range of values shown.

Note.--Selenium, vanadium, and uranium values for the limestone and alluvium/chinle aquifers are based on 4 to 5 analyses and must be regarded as tentative.

trations in mine effluents, together with low hydraulic heads, indicate short migration distances in groundwater for radionuclides and most stable trace elements in mine effluents.

Discharge of water pumped from mines to arroyos has both hydraulic and water quality impacts on shallow groundwater in the alluvial aquifer. The seepage model (Appendix H) and scattered field measurements in the Grants Mineral Belt substantiate that significant groundwater recharge is associated with mine discharge. Water quality effects on groundwater are poorly documented, however. We do not address the influence of impoundments used to remove suspended solids from mine effluents before discharge. Seepage water losses from such impoundments are believed to be small, especially when compared to infiltration losses in the arroyos and open fields receiving most of the wastes not piped to mills for process water. The impoundments are rather small and tend to become self-sealing due to settlement of fines. In at least one instance in Ambrosia Lake, the mine pond is lined to prevent seepage.

Unpublished flow and water quality data from the U.S. Geological Survey (P. Frenzel, written communication, 1979) document conditions in the Rio Puerco drainage near Churchrock and Gallup, New Mexico. Figure 3.21 shows the sampling station locations, and the chemical data are in Table 3.47. From October 1975 gaging data, seepage and evaporation reduce flow 9.39 m^3/min in a reach of 30.2 km, a loss of 0.31 m^3/min/km. Conservatively assuming 20 percent of this is by evaporation, seepage is 7.5 m^3/min or 3.94 x 10^6 m^3/yr. Gaging data for July 1977 and May 1978 similarly indicate average bed losses of 0.24 m^3/min/km. In the Ambrosia Lake district (data not shown), discharges (to San Mateo Creek and Arroyo del Puerto) from about a dozen mines total about 10.8 x 10^6 m^3/year, and the total length of perennial stream is about 15 kilometers. Assuming an average stream width of one meter and the above evaporation rate, evaporation and infiltration are 0.06 m^3/min and 7.54 m^3/min, respectively. In this case, infiltration amounts to 99 percent of total loss. Dissolved solids range from 520 to 1231 mg/ℓ (mean 743 mg/ℓ), and Ra-226 ranges from 0.2 to 23 pCi/ℓ (mean 6.6 pCi/ℓ). Selenium and molybdenum average 0.010 and 0.22 mg/ℓ, respectively.

Considering these two areas, evaporation averages about 4 percent of mine discharge versus the value of one percent calculated in Appendix H. Obviously, increased evaporation is accompanied by decreased infiltration.

Infiltration ranges from at least 90 percent to perhaps 99 percent of mine discharge, or from 1.8 to 1.98 m^3/min per mine. The foregoing field data and the more theoretical approach used in Appendix H show reasonable agreement on the relative amounts of infiltration and evaporation. We conclude then that most of the mine effluent infiltrates within relatively short distances of the mine(s) and recharges the shallow water table. The dissolved, generally nonreactive contaminants such as chloride and sulfate are expected to reach the water table, but reactive contaminants such as radium-226 and most trace metals would sorb or precipitate in the soil (substrate) in the course of infiltration.

The influence of mine discharge on groundwater quality beneath formerly ephemeral streams now receiving the discharge is currently under investigation by the New Mexico Environmental Improvement Division. Monitoring wells have been installed at several locations along the Rio Puerco (west) in the Churchrock area and San Mateo Creek in the Ambrosia Lake district. Table 3.50 summarizes partial results of samples taken in the last 12 to 18 months. In the Ambrosia Lake district, marked deterioration in water quality between the Lee Ranch and Sandoval Ranch stations on San Mateo Creek is a result of either natural causes and (or) mine drainage from a nearby deep underground uranium mine. Between Sandoval Ranch and Otero Ranch even more pronounced changes occur. In this short reach of 2.5 km, contaminated flows from uranium mines, ion-exchange plants, and seepage from an acid leach uranium mill enter Arroyo del Puerto, a tributary of San Mateo Creek. Additional study of surface water quality in the Arroyo del Puerto is recommended to further characterize the obviously interconnected surface water and groundwater systems.

In the Churchrock area, drained by the Rio Puerco, groundwater quality changes in the downstream direction are not readily apparent (Table 3.50). Although there is an acid leach mill also adjacent to the Rio Puerco tributary receiving the mine discharges, the mill is relatively new (1978 start-up) and may not yet influence stream quality. Most of the discharge from one of the two mines is used as mill feed water, thereby causing decreased discharge from the mines to the stream. Nevertheless, the reach of the perennial stream is increasing, indicating infiltration of remaining mine effluent and addition of water to storage in the shallow aquifer. Storage changes have been confirmed by static groundwater level measurements in the area east of

Table 3.50 Groundwater quality associated with the San Mateo Creek and Rio Puerco (west) drainages in the Grants Mineral Belt, New Mexico

Station	Sulfate (mg/ℓ)	Molybdenum	Selenium	Uranium
		(µg/ℓ)		
San Mateo Creek				
Lee Ranch	125.7	<10	< 5	<10
Sandoval Ranch	225-274	103-235	4-14.7	293-400
Otero Ranch	463-989	350-516	33-59	680-860
Rio Puerco (west)				
Hwy. 566 Bridge on N. Fork Rio Puerco	101-223	<10-284	20-22	530-760
Rio Puerco at Fourth St. Bridge, Gallup	163-244	<10-215	9-26	550-625

Source: Based on unpublished 1978 data developed by the New Mexico Environmental Improvement Division (J. Lazarus, oral communication, 1979).

Gallup. A massive spill of mill tailings into the Rio Puerco occurred in July 1979 and will complicate water quality investigation, insofar as the mine and mill influences are now superimposed in terms of both solid and liquid waste loadings in the watershed. The tailings "flood," estimated to contain about 360,000 m^3 of fluid and 1000 MT of solids, was traced into Arizona.

In summary and considering the high volume of dilute mine discharges, which are enriched in certain stable and radioactive toxic trace elements (EPA75; Hi77), we recommend that water quality effects of mine discharge be very carefully evaluated in at least a few selected areas. Available streamflow data indicate that infiltration is the principal means of disposal, yet the water quality data base, in particular, is rather weak to assess whether adverse impacts are likely. It is expected that future discharges in the Churchrock area alone will amount to about 40 m^3/min and will contain less than 400 mg/ℓ dissolved solids, most of which is sodium and bicarbonate. Dissolved concentrations of uranium, radium, iron, selenium, and vanadium are elevated relative to drinking water limits and infiltration of uranium, selenium, and possibly other stable elements warrants study. Use of settling ponds and barium chloride treatment greatly reduces the suspended solids, uranium, and radium concentrations. The final composition and ultimate disposal of pond sediments and added chemicals is essentially undocumented and bears additional investigation. Lastly, mine dewatering creates marked regional cones of depression and reduces the flow of water to existing supply wells and the baseflow component in major drainage systems such as the San Juan River (Ly79).

3.4.4 Gases and Dusts from Mining Activities

3.4.4.1 Radon-222 in Mine Exhaust Air

Unlike surface mines, large capacity ventilating systems are required in underground uranium mines, primarily to dilute and remove Rn-222 that emanates from the ore (Section 1.3.3). Ventilation rates vary from a few hundred to a few hundred thousand cubic meters of air per minute, and measured Rn-222 concentrations in mine vent air range from 7 pCi/ℓ to 22,000 pCi/ℓ (Ja79b). The concentration of Rn-222 in mine exhaust air varies depending upon ventilation rate, mine size (volume) and age, grade of exposed ore, size of active working areas, rock characteristics (moisture content and porosity), effectiveness of bulkhead partitions, barometric pressure, ore production rates, and mining practices. The emanation of Rn-222 dissolved in water that seeps into most mines may also contribute to Rn-222 in the exhaust air.

Because of the numerous variables that affect Rn-222 concentrations in mine air, it is difficult to confidently model radon releases from underground mines. A useful model would be one that would relate radon emissions to the production of U_3O_8. Measurements relating radon emissions to ore production have been made at seven underground uranium mines in New Mexico (Ja79b). The results of these measurements varied at the different mines from 1,380 to 23,500 Ci Rn-222 per AFR*, with an average rate of 4,300 Ci Rn-222 per AFR. The higher emission rates were noted to occur at the older mines. This was believed due to larger surface areas of exposed ore and sub-ore in the older mines. That is, inactive mined-out areas increase with mine age, and the ceiling, floors, and walls of these areas still contain certain amounts of ore and sub-ore. Radon emanating from these surface areas tend to increase the Rn-222 content of exhausted mine air unless these inactive areas of the mine (rooms, stopes, drifts, etc.) are effectively sealed. Because the radon emission factor is so variable in terms of Ci per AFR, an emission rate based on cumulative U_3O_8 mined has been proposed for modeling purposes (Ja79b). It is believed that this relationship would reduce the apparent dependence of the emission rate on the mine age. However, data are not presently available to make this latter correlation.

Although the average measured Rn-222 exhaust factor of 4,300 Ci/AFR is tentative and may be improved by studies in progress (Ja79b), it is the only value currently available for modeling purposes and will, therefore, be used in the present assessment. Assuming that 1 AFR is equivalent to 245 MT** of U_3O_8 (Ja79b), 0.017 Ci of Rn-222 will be released from the mine vents per metric ton of 0.1 percent grade ore mined. This emission rate will include all underground sources, i.e., emanation from exposed ore and blasting, slushing, loading, and transporting ore bearing rock. Radon-222 emissions were estimated for the two model underground mines by multiplying their respective annual ore capacities by the above emission rate. Table 3.51 lists the results.

---

*AFR = Annual Fuel Requirement for a 1000 MWe LWR.

**The AFR value on which the exhaust factor was based.

The estimated annual radon release computed for the average underground mine is compared below with releases reported elsewhere. Agreement is reasonably good.

Source	Annual Release of Rn-222, Ci
This Study	306
Tr79	289 - 467[(a)]
TVA78a	1577
TVA78b	180
TVA79	215
Th79	87

(a) Adjusted for 0.1 percent ore grade.

By properly capping the exhaust vents and sealing the shaft and mine entrance, radon emission rates from inactive mines will be a negligible fraction of the radon release rate that occurs during active mining.

3.4.4.2 Aboveground Radon-222 Sources

Radon-222 will be released from the following aboveground sources.

1. Dumping ore, sub-ore, and waste rock from the ore skip into haul trucks and unloading them on their respective piles.
2. Reloading ore from the stockpile after a 41-day residence time.
3. Emanation from waste rock, sub-ore, and ore storage pile surfaces.

The annual quantities of Rn-222 released by sources 1 and 2 were estimated using the following factors and assumptions.

- Radon-222 is in secular equilibrium with U-238.
- The density of ore, sub-ore, and waste rock is 2.0 MT/m^3.
- Annual production rates of ore and sub-ore are equal and assumed to be 1.8×10^4 MT at the average mine and 2×10^5 MT at the average large mine (Sections 3.4.1.1 to 3.4.1.3).
- The production rate ratio of ore to waste rock is 9.1:1 (Section 3.4.1.1).
- All Rn-222 present is available for release, 0.00565 Ci/m^3 per

Table 3.51 Estimated annual radon-222 emissions from underground uranium mining sources

Source	Average Mine[(a)], Ci/yr	Average Large Mine[(b)], Ci/yr
Underground		
Mine vent air	306	3,400
Aboveground		
Ore loading and dumping	1.4	15.3
Sub-ore loading and dumping	0.5	5.3
Waste rock loading and dumping	0.003	0.03
Reloading ore from stockpile	1.4	15.3
Ore stockpile exhalation	6.3	53
Sub-ore pile exhalation	61	338
Waste rock pile exhalation	0.5	2.6
Total	377	3830

(a) Annual production of ore and sub-ore = 1.8×10^4 MT, waste rock = 2.0×10^3 MT.

(b) Annual production of ore and sub-ore = 2×10^5 MT, waste rock = 2.2×10^4 MT.

percent U_3O_8 (Ni79), with an emanation coefficient of 0.27 (Au78, Tanner, A.B., Department of Interior, Geological Survey, Reston, VA., 11/79, personal communication). The quantities of U_3O_8 present in ore, sub-ore, and waste rock are 0.10 percent, 0.035 percent and 0.0020 percent, respectively (Sections 3.4.1.1 to 3.4.1.3).

Substituting the above values into the following equation yields the Rn-222 releases given in Table 3.51 for the average mine and the average large mine.

$$\text{Rn-222 (Ci/yr)} = (\text{percent } U_3O_8)\left(\frac{0.00565\ \text{Ci}}{\text{m}^3\ .\ \text{percent}}\right)(0.27)\left(\frac{\text{m}^3}{2.0\ \text{MT}}\right) \times \left(\text{Production Rate, } \frac{\text{MT}}{\text{yr}}\right) \qquad (3.11)$$

These releases are maximum values since very little time will have elapsed between the underground (blasting, slushing, loading, etc.) and surface operations. A significant amount of the radon that is available for release will emanate during the underground operations and invalidate the first assumption above concerning radioactive equilibrium. Nevertheless, these estimated maximum releases are very small in comparison to the radon released from the mine exhaust vents.

The emanation of Rn-222 from waste rock, sub-ore, and ore piles is based on an exhalation rate of 0.092 $\text{Ci/m}^2\cdot\text{yr}\cdot\text{percent } U_3O_8$ (Ni79) and ore grades of 0.002 percent, 0.035 percent, and 0.10 percent, respectively. Surface areas of the ore piles (Table 3.37), sub-ore piles (Table 3.38), and waste rock piles (Table 3.36) were used in these calculations. Applying these parameters, the annual Rn-222 emissions from the waste rock, sub-ore, and ore piles at the average mine and average large mine were computed. Table 3.51 gives the results. Total annual Rn-222 emissions during underground mining operations is the sum of the releases from all sources considered: 377 Ci from the average mine and 3830 Ci from the average large mine. More than 80% of the Rn-222 emissions results from the mine vent air.

3.4.4.3 Dusts and Fumes

Vehicular emissions resulting from the combustion of hydrocarbon fuels in gasoline and diesel-powered equipment are considerably less at underground mines than at surface mines (Section 3.3.4.1). The principal emissions are particulates, sulfur oxides, carbon monoxide, nitrogen oxides, and hydrocarbons. The quantity of these combustion products released to the atmosphere depends on the number, size, and types of equipment used, all of which are directly related to ore production.

EPA has estimated the following emissions from mining 1350 MT of ore per day from an underground mine (Re76).

Pollutant	Emissions per Operating Day, Kg/d
Particulates	2.4
Sulfur Oxides	5.0
Carbon Monoxide	41.9
Nitrogen Oxides	68.1
Hydrocarbons	6.9

Assuming a 330 operating-day year (Ni79), these emissions were adjusted according to the annual ore production of the average mine (1.8×10^4 MT) and the average large mine (2×10^5 MT). Table 3.52 lists the total airborne combustion product emissions. These emissions are small compared to those at surface mines (Table 3.30). For example, these estimates indicate that the emissions of combustion products at the average surface mine are more than 100 times greater than those at the average underground mine.

At underground mines, dust is produced by both underground and surface operations. No measurements have been made of dust concentrations in mine exhaust air. Because underground mines are wet, which greatly reduces dust production, and since a large portion of the dust produced would probably deposit underground, dust emissions from underground operations are probably relatively small. Hence, dust emissions from underground operations will not be assessed.

Aboveground sources of dust include dumping ore, sub-ore, and waste rock from the skip into haul trucks; dumping these materials onto their respective piles; reloading ore from the stockpile; using dirt haul roads by vehicular traffic; and dust suspended by the wind from the waste rock, sub-ore, and ore piles. These sources will be assessed as was done previously for surface mines (Section 3.3.4.1).

Table 3.52 Estimated air pollutant emissions from heavy-duty equipment at underground uranium mines

Pollutant	Emissions, Kg/yr[a] Average Mine[b]	Average Large Mine[c]
Particulates	32	350
Sulfur oxides	67	740
Carbon monoxide	560	6,210
Nitrogen oxides	910	10,100
Hydrocarbons	92	1,020

(a)Based on Re76 and 330 operating days per year.

(b)Annual ore production = 1.8 x 10^4 MT.

(c)Annual ore production ≐ 2 x 10^5 MT.

Dust emissions will vary over a wide range depending upon moisture content, amount of fines, number and types of equipment operating, and climatic conditions. Because ore is generally wet, the relative amounts of dust produced from its mining and handling are usually small. The following emission factors were selected from those suggested by the EPA for loading and dumping operations (Hu76, Ra78, Da79):

truck loading = 2.5 x 10^{-2} kg/MT; and
truck dumping = 2.0 x 10^{-2} kg/MT.

Average annual dust emissions were estimated for the aboveground mining activities by applying these emission factors to the ore, sub-ore, and waste rock production rates of the average mine and average large mine. Table 3.53 lists the results. One-half the emission factor values were applied to ore and sub-ore because they are generally wet, except when reloading ore from the stockpile. In that case, it is assumed to have dried during the 41-day residence period (Section 3.4.1.2). Also, the emission factor for truck loading was assumed valid for loading the haul trucks from the mine skip. The dust emission for truck dumping may be high since it was based on dumping of aggregate, which would have a smaller particle size distribution than the ore, sub-ore, or waste rock (Hu76).

The movement of heavy-duty trucks is a large source of dust at most uranium mines. The magnitude of this source depends upon a number of factors, including the particle size distribution and moisture content of the road bed material, vehicular speed and distance traveled, and meteorological conditions. Emission factors for heavy-duty haul trucks (1.15 kg/VKmT) and light duty vehicles (1.03 kg/VKmt) are the same as those computed for these vehicles at surface mines (Section 3.3.4.1). Dust emissions for the movement of heavy-duty haul trucks were estimated using the appropriate emission factor and assuming --

- 31.8 MT truck capacities;
- round-trip haul distances of 1.61 km to the ore and sub-ore piles and 3.22 km to the waste rock pile; and
- the annual production rates given in Sections 3.4.1.1, 3.4.1.2 and 3.4.1.3.

Table 3.53 lists the results.

Additional dust emissions will occur from light-duty vehicular traffic along access roads. Using the emission factor derived in Section 3.3.4.1 (1.03 kg/VKmt) and assuming that there are 16 km of access roads traveled 4 times a day during the 330 operating days per year, about 22 MT of dust will be produced from this source annually. Emissions that occur during haulage road maintenance is relatively small and will not be considered.

Heavy-duty, haul truck traffic at underground uranium mines produces considerably less dust than at surface mines. This is to be expected because of the vast quantities of overburden that must be transported as well as larger ore and sub-ore capacities at surface-type mines.

The dust emissions computed above for transportation assume no effective dust control program. But, haul roads are normally sprinkled routinely during dry periods, and stabilizing chemicals are applied to roadways, usually to the ore haul roads. Dust emissions along haul roads can be reduced by 50 percent from sprinkling and up to 85 percent by the application of stabilizing chemicals (EPA77b, Da79).

Table 3.53 also lists average annual dust emissions caused by wind erosion of waste rock, sub-ore, and ore piles at the model underground mines. Emission factors, computed in Appendix I, are 2.12 MT/hectare-yr for waste rock and sub-ore piles and 0.040 kg/MT for the ore stockpiles. The first emission factor was multiplied by the waste rock and sub-ore pile surface

Table 3.53 Estimated average annual dust emissions from underground mining activities

Source(d)	Dust Emissions, MT/yr					
	Average Mine(a)			Average Large Mine(b)		
	Ore(c)	Sub-ore(c)	Waste Rock	Ore(c)	Sub-ore(c)	Waste Rock
Loading truck from skip at mine shaft	0.23	0.23	0.05	2.5	2.5	0.6
Truck dumping at piles	0.18	0.18	0.04	2.0	2.0	0.4
Reloading stock-piled ore(e)	0.45	NA(f)	NA	5.0	NA	NA
Wind suspended dust from piles	0.72	4.0	0.57	8.0	22	3.0
Transportation(g)	1.0	1.0	0.23	11.6	11.6	2.6

(a) Based on annual production rates of 1.8×10^4 MT of ore and sub-ore, and 2.0×10^3 MT of waste rock.

(b) Based on annual production rates of 2×10^5 MT of ore and sub-ore, and 2.2×10^4 MT of waste rock.

(c) Assumed wet.

(d) Aboveground activities.

(e) Assumed dry.

(f) NA - Not applicable.

(g) Dust emissions from heavy-duty, vehicular traffic along ore, sub-ore, and waste rock haul roads.

areas given in Tables 3.36 and 3.38, respectively, while the second factor was multiplied by the annual ore production.

Table 3.54 shows annual contaminant emissions caused by mining activities (loading and dumping) according to source location, at the mine shaft and at the piles. Contaminant emissions were computed by multiplying the total annual dust emissions at each pile (Table 3.53) by the respective contaminant concentrations in each source--waste rock (Section 3.4.1.1; Table 3.16), sub-ore (Section 3.4.1.3; Table 3.19), and ore (Section 3.4.1.2; Table 3.19). Contaminant emissions at the site of the mine shaft were computed by multiplying the annual dust emissions of ore, sub-ore, and overburden (loading truck from skip - Table 3.53) by their respective contaminant concentrations. The three products of the multiplication were then summed to give the values listed in the 4th and 8th data columns of Table 3.54. The health impact of the sources at each location will be assessed separately in Section 6.1.

Annual contaminant emissions due to wind suspension and transport of dust are listed in Table 3.55. These values were computed by multiplying the annual mass emissions (Table 3.53) by the contaminant concentrations in waste rock, sub-ore, and ore listed in Sections 3.4.1.1, 3.4.1.3, and 3.4.1.2, respectively. The uranium and uranium daughter concentrations in dusts from all sources were also multiplied by an activity ratio (dust/source) of 2.5 (Section 3.3.1.2). Although some metals may also be present as secondary deposits, it was believed that there were insufficient data to justify multiplying their concentrations by the 2.5 ratio.

The dust emissions from vehicular traffic listed in Table 3.53 (transportation) were summed with that produced by light vehicular traffic (22 MT/yr) and considered one source of emissions. Concentrations of contaminants in haul road dust have not been measured and are not known. Some spillage of ore and sub-ore along haul roads will undoubtedly raise uranium levels in roadbed dust. As an estimate, uranium and daughter concentrations in the dust were considered to be twice background, 8ppm (2.7 pCi/g), while concentrations of all other contaminants were considered to be similar to those in the waste rock (Section 3.4.1.1). Table 3.56 shows the annual emissions computed with these assumptions.

Table 3.54 Average annual emissions of radionuclides (μCi) and stable elements (kg) from mining activities at the model underground mines

	Average Underground Mine[a]				Average Large Underground Mine[a]			
Contaminant	Waste Rock Pile Site	Sub-ore Pile Site	Ore Pile Site	Mine Site	Waste Rock Pile Site	Sub-ore Pile Site	Ore Pile Site	Mine Site
Arsenic	0.0004	0.02	0.05	0.04	0.004	0.17	0.60	0.44
Barium	0.012	0.17	0.58	0.44	0.12	1.8	6.4	4.8
Cobalt	NR[b]	0.003	0.01	0.007	NR	0.03	0.11	0.08
Copper	0.0007	0.01	0.04	0.03	0.007	0.12	0.43	0.32
Chromium	< 0.002	0.004	0.01	<0.01	< 0.02	0.04	0.14	< 0.13
Iron	0.24	2.8	9.9	7.5	2.4	3.1	110	82
Mercury	< 0.0003	ND[c]	ND	<0.001	< 0.003	ND	ND	0.005
Potassium	0.28	4.5	16	12	2.8	50	175	129
Magnesium	NR	0.63	2.2	1.6	NR	7.0	25	18
Manganese	0.02	0.17	0.60	0.47	0.19	1.9	6.7	5.1
Molybdenum	0.0001	0.02	0.07	0.05	0.001	0.23	0.81	0.58
Nickel	NR	0.004	0.01	0.009	NR	0.04	0.14	0.10
Lead	0.0009	0.01	0.05	0.04	0.009	0.16	0.55	0.40
Selenium	0.0001	0.02	0.07	0.05	0.001	0.22	0.77	0.55
Strontium	0.006	0.02	0.08	0.07	0.06	0.26	0.91	0.74
Vanadium	0.004	0.25	0.89	0.65	0.04	2.8	9.9	7.1
Zinc	0.0008	0.005	0.02	0.01	0.008	0.06	0.20	0.16
Uranium-238 and each daughter	0.6	45	450	222	6	495	4,990	2,410
Thorium-232 and each daughter	0.04	0.4	6.3	2.8	0.4	4	70	31

(a)Mass emissions from Table 3.53.

(b)NR - Not reported.

(c)ND - Not detected.

Table 3.55 Average annual emissions of radionuclides (μCi) and stable elements (kg) in wind suspended dust at the model underground mines

Contaminant	Average Large Underground Mine			Average Underground Mine		
	Waste Rock Pile	Sub-Ore Pile	Ore Stockpile	Waste Rock Pile	Sub-Ore Pile	Ore Stockpile
Arsenic	0.03	1.9	0.69	0.005	0.34	0.06
Barium	0.87	20	7.4	0.17	3.7	0.66
Cobalt	NR[(a)]	0.35	0.13	NR	0.06	0.01
Copper	0.05	1.3	0.49	0.01	0.24	0.04
Chromium	<0.15	0.44	0.16	< 0.03	0.08	0.01
Iron	18	345	126	3.4	63	11
Mercury	<0.02	ND[(b)]	ND	< 0.005	ND	ND
Potassium	21	550	200	4.0	100	18
Magnesium	NR	77	28	NR	14	2.5
Manganese	1.5	21	7.7	0.28	3.8	0.69
Molybdenum	0.008	2.5	0.92	0.001	0.46	0.08
Nickel	NR	0.44	0.16	NR	0.08	0.01
Lead	0.07	1.7	0.62	0.01	0.31	0.06
Selenium	0.006	2.4	0.88	0.001	0.44	0.08
Strontium	0.45	2.9	1.0	0.09	0.52	0.09
Vanadium	0.30	31	11	0.06	5.6	1.0
Zinc	0.06	0.64	0.23	0.01	0.12	0.02
Uranium-238 and each daughter	45	5,450	5,700	9	990	513
Thorium-232 and each daughter	3	44	80	0.6	8	7.2

(a) NR - Not reported.

(b) ND - Not detected.

Table 3.56 Average annual emissions of radionuclides (μCi) and stable elements (kg) from vehicular dust at the model underground mines

Contaminant	Average Large Underground Mine[(a)]	Average Underground Mine[(b)]
Arsenic	0.43	0.22
Barium	14	7.0
Copper	0.86	0.44
Chromium	<2.4	<1.2
Iron	287	145
Mercury	<0.38	<0.19
Potassium	335	170
Manganese	23	12
Molybdenum	0.12	0.06
Lead	1.1	0.53
Selenium	0.10	0.05
Strontium	7.2	3.6
Vanadium	4.8	2.4
Zinc	0.96	0.48
Uranium-238 and each daughter	129	65
Thorium-232 and each daughter	48	24

(a) Mass emissions = 47.8 MT/yr.

(b) Mass emissions = 24.2 MT/yr.

3.5 In Situ Leach Mining

Because in situ leaching of uranium (see general description in Section 1.3.4) is in its infancy, a data base for performing a detailed generic environmental assessment does not presently exist. The fact that the parameters for assessing this process are so site specific and depend upon operational procedures further impedes a generic assessment. Current research projects may help to resolve many of the present uncertainties and provide the data needed to better quantify the potential source terms (La78).

In view of the expected future expansion of this uranium mining method (Section 1.3.4), a qualitative assessment that can be modified later when additional data become available was deemed necessary. This assessment was possible because of recent laboratory experiments and field measurements at pilot-scale plants (Wy77, Ka78b, NRC78b, Tw79).

Similar to other uranium mining methods, in situ leaching also produces liquid, solid, and airborne wastes. However, the quantities of these wastes and their characteristics differ considerably from those produced at surface or underground mines. Also, because the recovery, drying, and packaging of the U_3O_8 produced is often performed at the mine site, wastes from these processes should probably be included in the mine assessment.

This assessment uses the parameters of a hypothetical "typical" commercial-sized in situ solution mine. Unlike surface or underground mines, relatively few in situ facilities exist, and they are all somewhat different because of site specificity and the rapid development of new or modified techniques. The following parameters for the hypothetical mine were based upon those of the Highland, Crownpoint, and Irigaray uranium projects and those reported by Kasper et al. (1978) (Wy77, NRC78b, TVA78b).

The Hypothetical In Situ Solution Mine

(1) Size of deposit = 52.6 hectares
(2) Average thickness of ore body = 8 m (Ka78b, NRC78b)
(3) Average ore grade = 0.06 percent U_3O_8 (Ka78b, Tw79)
(4) Mineralogy = Sandstone

(5) Ore density = 2 MT/m^3

(6) Ore body depth = 153 m

(7) Mine life = 10 years (2-yr leach period in each of 5 sectors)

(8) Well pattern = 5 spot (NRC78b, TVA78b, Ka78b)
Injection wells = 260
Production wells = 200
Monitoring wells = 80

(9) Annual U_3O_8 production = 227 MT (Wy77, NRC78b, Ka78b)

(10) Uranium leaching efficiency = 80 percent (Ka78b)

(11) Lixiviant = Alkaline

(12) Lixiviant flow capacity = 2,000ℓ/min (Ka78b, Wy77, NRC78b)

(13) Lixiviant bleed = 50ℓ/min (2.5 percent) (Wy77, NRC78b, TVA78b)

(14) Uranium in Lixiviant = 183 mg/ℓ (TVA78b, Ka78b, NRC78b)

(15) Calcite ($CaCO_3$) removal required = 2 kg calcite per kg U_3O_8 (Wy77)

The solid, liquid, and airborne wastes generated by this facility are described below. Wastes and quantities generated, as well as operations and procedures selected, will naturally differ to varying degrees from those at some operating sites.

3.5.1 Solid Wastes

The quantity of solid wastes generated depends upon the leachate, the ore body, and operational procedures that effect the mobilization of ore constituents. Little information is available on the quantities of solids generated because of this site dependence, the newness of the process, and the apparent relatively small quantities that are produced. Examples of solid wastes that might be expected to be generated by the alkaline leach process are listed below:

(1) Materials filtered from the lixiviant line

(2) Sediments from the surge tanks

(3) Calcium carbonate from the calcium control unit

(4) Barium sulfate from the contaminant control in the elution/precipitation circuit of the recovery process
(5) Materials deposited in the evaporation ponds
(6) Drill hole residues
(7) Solids from aquifer restoration

Sources 1 and 2

No information concerning quantities of solids from these two sources could be found in the literature, but they are described as being relatively small compared to other sources (NRC78b). These wastes are transferred to evaporation ponds and retained beneath a liquid seal.

Source 3

One of the larger sources of solids is the calcium control unit (Wy77). Calcite, $CaCO_3$, which is removed prior to injection of the refortified lixiviant, coprecipitates radium and any residual uranium. It has been reported that the amount of calcite produced is less than 2.8 kg per 1 kg of U_3O_8 recovered (Wy77). Assuming this ratio to be 2.0, and if Ra-226 is in secular equilibrium with U-238 in the ore, and 2.5 percent is solubilized by the lixiviant (Wy77, NRC78b), 454 MT of calcite will be produced annually and contain a total of 1.6 Ci of Ra-226. Also, calcite has been observed to contain between 1 to 2 percent U_3O_8 by weight (Wy77). Assuming an average of 1.5 percent U_3O_8, about 1.9 Ci (6.8 MT U_3O_8) of U-238 may also be present in the calcite waste.

Radium-226 and its daughter, Rn-222, are probably the most radiologically significant radionuclides associated with uranium mine wastes, and the small amount of Ra-226 retrieved by in situ leaching is a distinct advantage. Conventionally mining the quantity of ore assumed for the hypothetical in situ mine would contribute 64 Ci of Ra-226 per year to the surface. Because of the insolubility of $RaSO_4$, acid lixiviants containing H_2SO_4 mobilize even less radium than alkaline lixiviants. It is reported that the latter mobilizes up to 4.5 times the radium as acid leach solutions (Wy77).

If practical, the calcite waste is transferred to the mill to recover the coprecipitated uranium. Otherwise, the waste is transferred to an evaporation pond and retained beneath a liquid seal to minimize atmospheric dispersion and radon emanation.

Source 4

If necessary, the sulfate concentration in the eluant circuit of the uranium recovery unit may be controlled by the precipitation of $BaSO_4$. There are no data on the contaminant levels expected in the $BaSO_4$ waste, although less than 730 MT per year are anticipated (Wy77). These wastes are impounded beneath a liquid seal of an evaporation pond.

Source 5

An assortment of precipitation compounds will be produced by evaporative concentration of impounded waste solutions. The principal products expected are alkali chlorides, carbonates, and sulfates. The quantity of solids produced by this mechanism and their rate of accumulation on the pond bottom has not been reported.

Source 6

Residues produced from drilling the numerous wells required for in situ leaching constitute another solid waste. The hypothetical in situ leaching facility defined above requires a total of 540 wells drilled to a depth of 153 m: 200 production, 260 injection, and 80 monitoring wells. A diameter of 10.2 cm will be assumed for all wells, although 5.1 cm, 12.7 cm, and 15.2 cm diameter wells have been used (Wy77). To accommodate a concrete and steel casing, a drill hole of approximately 20 cm will be required. The residue from drilling the monitoring wells will consist mostly of barren rock; however, an equivalent of an 8-m section of each injection and recovery well will contain 0.06 percent grade ore. Hence, drill hole residues will consist of 4,960 MT of barren waste rock and 230 MT of ore containing 138 kg of U_3O_8. These wastes are in relatively small quantities and should be manageable. The waste rock and ore, if mixed and stored in a 2-m-high rectangular pile, would only cover an area of about 0.15 hectares and average 0.0027 percent U_3O_8.

Source 7

During the active mining period, all solid wastes are generally retained beneath a liquid seal in lined evaporation ponds to minimize atmospheric dispersion and radon emanation. A plan for the final disposal of solid wastes has not been determined. Suggested procedures are to transport the wastes to a conventional uranium mill for further treatment to recover any U_3O_8 present, treat the effluent as mill wastes, construct long-term tailings ponds on the site, or ship the wastes to a licensed off-site burial ground. Solid wastes probably comprise the least significant type waste relative to health and the environment. Solid wastes generated from reclamation procedures will be discussed in Section 3.5.5.

3.5.2 Associated Wastewater

Water flushed through the leached area when restoring the well field is the largest source of wastewater (see Section 3.5.5). The principal sources of wastewater generated by the hypothetical facility during the leaching and recovery operations are as follows:

(1) Lixiviant bleed -- barren lixiviant removed from the leach circuit to produce a net inflow into the well-field area and to control contaminant concentrations

(2) Resin wash -- water to wash resin of excess NH_4Cl used to regenerate the resin. Lixiviant bleed is sometimes used for this operation, and it reduces the total quantity of wastewater produced (Ka78b)

(3) Eluant bleed -- barren eluant removed to control salt accumulation, principally NaCl and Na_2CO_3, and maintain proper volume

(4) Well cleaning -- water used to flush injection wells to prevent clogging

The sources of wastewater and the quantities produced vary at different sites, depending upon the lixiviant and recovery circuit chemistry as well as the production rates. However, estimates were made of the quantities of wastewater generated by the four principal sources for the hypothetical in situ facility, and they are listed in Table 3.57. It is assumed that waste from backwashing the sand filters is lixiviant bleed waste water and does not contribute to the total wastewater generated. The total volume of wastewater estimated to be generated is 8.43×10^4 m^3/yr. Assuming the evaporation ponds are 3.05 m deep with a 0.604 m freeboard (Wy77) and a natural evaporation rate of 142 cm/yr (TVA78b), a pond capacity of 34,770 m^3/yr which would encompass a surface area of about 1.4 hectares/yr would be required. Using evaporation data assumed for the Irigaray Uranium Project, about 75 percent of the annual wastewater inventory would evaporate, which would leave 2.11×10^4 m^3/yr and require a surface area of 0.85 hectares/yr. If necessary, the pond size can be reduced by using mechanical evaporators.

Table 3.57 Estimated quantities of wastewater produced by an in situ leaching operation

Source	Flow Rate, (ℓ/min)	Annual Accumulation, (m^3/yr)
Lixiviant bleed (2.5%)	50	2.63×10^4
Resin wash[a]	26	1.37×10^4
Eluant bleed	17	8.9×10^3
Well cleaning[b]	--	3.54×10^4
Total		8.43×10^4

[a] This may be included in the lixiviant bleed.

[b] Assumes 260 injection wells flushed twice each month with 5680 liters of water.

Source: Data from Wy77 and Ka78b proportioned to an annual U_3O_8 production of 227 MT and a lixiviant flow of 2000 ℓ/min; aquifer restoration is excluded (Section 3.5.5).

The liquid wastes are generally brines. They contain large amounts of sodium chloride consisting of 1,500 to 5,000 mg/ℓ total dissolved solids (TDS), trace metals ranging from 0 to 10 mg/ℓ, and small quantities of radioactivity. The quantities of contaminants generated each year were estimated for the hypothetical solution mine by using the annual mass emissions estimated for the Highland Uranium Project and adjusting the flow rates to predict the concentrations (NRC78b). Table 3.58 lists these estimated concentrations and annual emissions. Because the contaminants from the lixiviant bleed were not included in the source document, the trace metals that are mobilized by the leachate do not appear in the tabulation, and Ra-226 presence is grossly underestimated (Table 1.7, Section 1.3.4). Considering possible trace metal concentrations and their toxicities, their presence in the lixiviant bleed wastewater may be significant. Assuming that 2.5 percent of the Ra-226 in the ore is extracted, the pregnant leachate will contain about 1,520 pCi/ℓ, yielding 1.6 Ci/yr. However, it is assumed that most of this radium will be removed by the calcium control unit.

There are no planned releases of liquid wastes to the environment at in situ solution mines. The contaminants dissolved in the liquid wastes will accumulate on the pond bottoms as the liquid evaporates. Barring dike failure and seepage through the lined pond bottoms, no impact should be imposed upon the environment by this source during operation.

Another method, other than evaporation, to remove wastewater from an in situ site is deep well injection. This is the dominant method of wastewater removal at operations in South Texas (Durler, D.L., U.S. Steel Corporation, Texas Uranium Operations, Corpus Cristi, TX, 9/79, written communication).

3.5.3 Airborne Emissions

Airborne emissions from an in situ solution mining operation will originate from three principal sources: the uranium recovery and processing unit, the waste storage evaporation ponds, and the radon released from the pregnant leach surge tanks. The primary radioactive species emitted is Rn-222. The nonradioactive species emitted are a function of the lixiviant and the uranium recovery processes employed. Fugitive dust emissions, primarily from vehicular traffic, will also occur on the site. However, because very little heavy equipment is used, the potential for adverse environmental impact from this source will not be significant and is not considered in this assessment.

Table 3.58 Estimated average concentrations and annual accumulation of some contaminants in wastewater

Contaminant	Concentration, mg/ℓ	Annual Accumulation, kg
Calcium	64	5,380
Chlorine	2,070	173,880
Carbonate	31	2,600
Bicarbonate	36	3,020
Magnesium	24	2,020
Sodium	1,320	110,880
Uranium-238	1	84
Radium-226	21(a)	1.8(b)
Thorium-230	6(a)	0.5(b)

(a)Units are pCi/ℓ

(b)Units are mCi.

Note.--Mass emissions estimated for the Highland Uranium Project (NRC78b), adjusted for flow rates and U_3O_8 production of the hypothetical solution mine.

Estimated average annual airborne emissions were computed for the hypothetical facility using data supplied by the Irigaray and Highland Uranium Projects and from the report of Kasper, et al. (1978) (Wy77, NRC78b). Table 3.59 gives the results, proportioned to a production rate of 227 MT/ yr.

The major sources of emissions from the uranium recovery plant are by-products of combustion from the dryers, volatilized solution residuals, and U_3O_8 fines generated during product drying. Carbon dioxide is the major combustion product emitted, although sulfur dioxide may also be significant if oil is used to fuel the dryers. Ammonium salts, used in the precipitation of uranium and resin regeneration, will volatilize as both ammonia and ammonium chloride during yellow cake drying. Airborne particulates that include uranium and some decay products are generated during the drying and packaging processes. The emission rates of U_3O_8 and daughter products were computed on the basis of an average release rate of 363 kg of U_3O_8 per year

Table 3.59 Estimated average annual airborne emissions from the hypothetical in situ leaching facility

Source	Annual Release Rate
Recovery Plant[a]	
Uranium-238	1.0×10^{-1} Ci
Uranium-234	1.0×10^{-1} Ci
Uranium-235	4.8×10^{-3} Ci
Thorium-230	1.7×10^{-3} Ci
Radium-226	1.0×10^{-4} Ci
Lead-210	1.0×10^{-4} Ci
Polonium-210	1.0×10^{-4} Ci
Ammonia	3.2×10^{0} MT
Ammonium chloride	1.2×10^{1} MT
Carbon dioxide	6.8×10^{2} MT
Surge Tank	
Radon-222[b]	6.5×10^{2} Ci
Storage Ponds[c]	
Ammonia	1.0×10^{2} MT
Ammonium chloride	3.0×10^{2} MT
Carbon dioxide	7.5×10^{1} MT

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[a] Includes the calcium control unit.

[b] Assumes all radon formed dissolves in the lixiviant and 100 percent is released on contact with the atmosphere.

[c] Based on a release rate of 14.6 MT/yr of NH_3, 10.6 MT/yr of CO_2 and 42.0 MT/yr of NH_4Cl per hectare of pond surface (Wy77), and an average pond surface area of 7.1 hectares (1.42 ha/yr x 5 yrs).

from a 227 MT/yr facility (Wy77, Ka78b). High efficiency filters and scrubbers are used, which significantly reduce the releases from the uranium recovery plant.

Emission rates from the wastewater storage ponds are determined by the composition of the waste solutions, evaporation rate, feed rate to the ponds, and the water temperature. The principal emissions from storage ponds servicing an alkaline leach process, as defined for the hypothetical facility, are ammonia, ammonium chloride, and carbon dioxide. Different atmospheric releases would result from waste ponds servicing an acid leach facility. The release of Rn-222 from the pond surfaces has not been measured. The emission rate of Rn-222 resulting from the decay of Ra-226 contained in the pond sediments will be inhibited by the liquid seal maintained over the entire surface area of the pond. Because of its low solubility in the unagitated pond water, it is reasonable to conclude that the rate of release for radon from the water surface will be small compared to that from the pregnant leach surge tanks. The liquid seal maintained over the pond area minimizes airborne particulate emissions from the storage ponds.

The principal source of airborne radioactive emissions is the release of Rn-222 from the pregnant leach surge tanks. Rn-222 is mobilized from the ore zone during solution mining and will be largely soluble in the lixiviant under the very high pressure (~15 atm) that exists at the ore zone depth (~500 ft). Upon reaching the atmosphere at the surge tank, nearly complete release of the absorbed radon will take place. Since nearly all Ra-226 remains underground in the leach zone--only 2.5 percent is assumed to be extracted--Rn-222 will continue to be generated in areas leached of uranium.

Consider a 2-year leach period in each of 5 sectors that is 80 percent efficient and yields an average of 227 MT of U_3O_8 per year. If U-238 and Ra-226 are initially in secular equilibrium and 97.5 percent of the Ra-226 remains underground, 156 Ci of Ra-226 will be continually available for Rn-222 production. This quantity of Ra-226 will yield a lixiviant concentration in the 252,800 m^3 aquifer (Section 3.5.5) of 6.18×10^5 pCi/ℓ, assuming a maximum emanating power of 100 percent. The latter assumption will result in a maximum Rn-222 concentration in the lixiviant. A high emanating power is probable considering the conditions that exist in the aquifer: high pressure, high permeability due to leaching, the presence of water in the rock pores, radium present on grain surfaces, and the flow rate of water through the ore zone (Ta78, Tanner, A.B., Department of Interior, Geological Survey, Reston, Va, 11/79, personal communication). Therefore, applying these maximizing conditions with a pumping rate of 2,000 ℓ/min, 650 Ci/yr of Rn-222 will be released at the pregnant leachate surge tanks.

Apparently very few measurements of Rn-222 concentrations in pregnant leachates have been made at operating facilities. One investigator reports that measured concentrations range from 10,000 pCi/ℓ to over 500,000 pCi/ℓ and may vary with time at the same well by factors greater than ten (Waligora, S., Eberline Instrument Corp., Albuquerque, N.M., 1979, personal communication). The concentration computed above for the model facility lies above the observed range.

3.5.4 Excursion of Lixiviant

A production zone excursion refers to the event when the leach solution flows from the leach field contaminating the surrounding aquifer. Production zone excursions are usually prevented by bleeding a small fraction (2 to 7 percent) of the lixiviant before reinjection. This imposes an imbalance in the injection-recovery volumes and causes groundwater to flow into the leach field from the surrounding stratum.

Production zone excursions are detected by wells placed 60 m to 300 m from the well field. These wells are routinely monitored, generally biweekly, to detect concentration increases of one or more constituents of the lixiviant. Lixiviant constituents monitored may be chloride, ammonia, bicarbonate, sulfate, calcium, or uranium. In addition, conductivity and pH measurements are usually included. When one or more of the indicators exceeds a maximum limit specified in the operator's permit, the observation is verified by resampling. If positive, sampling frequency is increased, appropriate government agencies are notified and corrective actions are begun.

An excursion from the production zone may be terminated by one of the following suggested methods (Wy77):

(1) Overpumping - increasing the flow rate of the recovery wells to increase the inward flow of native groundwater

(2) Reordering - applying different pumping rates of the recovery wells to different areas of the well field, providing a greater inflow of native groundwater at specific points

(a variation of overpumping)

(3) Reducing Injection - another method of increasing the ratio of recovery flow to injection flow providing the same effect as overpumping

(4) Ceasing to Pump - stopping both recovery and injection flows (migration is then due entirely to natural groundwater flow, which is many orders of magnitude less than with wells pumping)

(5) Begin Restoration - initiated when all other efforts have failed to stop the migration of lixiviant from the leach field (Section 3.5.5)

Excursions are likely to occur during the operation of an in situ leach mine. Adverse consequences of an excursion will be determined by its extent, the rate of outward flow, contamination levels, aquifer hydrology, and the effectiveness of corrective measures applied.

3.5.5 Restoration and Reclamation

Restoration is the process by which the in situ leach site is returned to an environmentally acceptable state after mining is complete. Surface restoration consists of removing all structures, pipelines, and so on and sealing the evaporation ponds. Subsurface restoration, the primary area of concern, is done by discontinuing lixiviant injection and continuing pumping to sweep fresh groundwater from the surrounding area through the leached ore zone. It is anticipated that this process will flush out the remaining lixiviant and chemical compounds or elements that have adsorbed or reacted with the mineral content of the aquifer. The water recovered can be purified by chemical precipitation, ion exchange, reverse osmosis, or other processes, and then recycled. This reduces considerably the quantity of water that must be managed. Between 75 and 80 percent of the water can be reinjected while the remainder containing the contaminants is transferred to an evaporation pond (Wy77, NRC78b). During the initial restoration process, it is generally cost effective to recover the uranium from the process wastewater.

Aquifer restoration continues until the groundwater quality in the mining zone meets a criterion established on a basis of the premining water quality. In many cases, the premining groundwater quality criterion is difficult to establish because water quality can vary considerably over the ore zone region and may contain high natural levels of contaminants. Samples of water from wells monitored prior to mining in Texas contained concentrations of Rn-222 approaching 20,000 pCi/ℓ (Tanner, A.B., Department of Interior, Geological Survey, Reston, Va, 11/79, personal communication), and it is probably unrealistic to attempt to restore an aquifer to a better quality than existed naturally before mining. Wells and flow rates used in this process must be carefully selected and controlled to provide efficient groundwater sweeps and to insure that all affected areas of the leach zone are restored.

The affected aquifer volume that is to be restored may be estimated by the following equation:

$$\text{affected volume} = \text{area of well field} \times \text{aquifer thickness} \times \frac{(\text{porosity})}{100 \text{ percent}} . \qquad (3.12)$$

Assuming a porosity for sandstone of 30 percent (NRC78b), the affected volume of the hypothetical in situ solution mine defined in Section 3.5 would be:

$$\text{affected volume} = 52.6 \text{ hectares} \times 8 \text{ m} \times 30 \text{ percent}/100 \text{ percent} = 1.26 \times 10^6 \text{ m}^3 .$$

Because of mixing leach solution with the incoming sweep water and the gradual desorption of some contaminants from clays present in the ore body, more water is required to adequately flush the contaminants than one pore volume. It has been estimated that five to seven pore volumes of water would be required for adequate restoration (Wy77, NRC78b). Using the seven pore volume value and assuming that 80 percent of the sweep water is reinjected after purification, a total of 1.76×10^6 m^3 of wastewater having high TDS would be transferred to the evaporation ponds during the restoration phase. If the aquifer is swept at a flow rate of 2,000 ℓ/min, restoration would take 8 years (1.6 yr per sector), and wastewater will accumulate at about 2.22×10^5m^3/yr during this period. With careful control, restoration can be concurrent with leaching in different areas of the well field.

Table 3.60 lists estimated average concentrations of contaminants in the restoration wastewater (NRC78b) and annual accumulation rates of the contaminants based on a flow rate of 2,000 ℓ/min. In the last column are estimates of the total mass of substances produced by restoration that would become sediments in the evaporation ponds. Data were not provided for calcium, magnesium, chloride, and ammonium ions, even though the latter two are major constituents expected from an alkaline leach process (Wy77). These concentrations reflect average values, but concentrations in the wastewater during the initial phase of the restoration process will be much higher. For

Table 3.60 Estimated average concentrations and annual and total accumulations of some contaminants in restoration wastewater

Contaminant	Concentration mg/ℓ	Annual Accumulation, Kg[a]	Total Accumulation, MT[b]
Arsenic	0.2	210	1.7
Calcium	NA[c]	NA	NA
Chloride	NA	NA	NA
Carbonate	450	473,000	3,780
Bicarbonate	550	578,000	4,620
Magnesium	NA	NA	NA
Sodium	550	578,000	4,620
Ammonium	NA	NA	NA
Selenium	0.10	100	0.8
Sulfate	150	157,000	1,250
Uranium-238	< 1[d]	< 900	< 7.2
Thorium-230	100[e]	0.10[f]	0.8[f]
Radium-226	75[e]	0.08[f]	0.6[f]
Radon-222	618,000[e]	650[f]	5,200[f]

[a] Produced only during the estimated 8-yr restoration period.

[b] Total accumulation during the estimated 8-yr restoration period.

[c] NA - Data not available.

[d] Concentration after uranium extraction.

[e] Units are pCi/ℓ.

[f] Units are Ci/yr or total curies.

Source: Concentrations based on those estimated for the Highland Uranium Project (NRC78b), adjusted for a flow rate of 2,000 ℓ/min.

Table 3.61 A comparison of contaminant concentrations in pre-mining groundwater and pre-restoration mine water (Wy77)

Contaminant	Pre-mining Water, mg/ℓ	Pre-restoration Water, mg/ℓ
Arsenic	<0.0025	0.021
Barium	0.12	0.069
Boron	0.16	0.283
Cadmium	<0.005	0.014
Chromium	0.0135	0.002
Copper	0.019	0.220
Manganese	0.12	0.97
Mercury	0.0028	<0.0002
Nickel	0.018	0.218
Selenium	0.013	1.75
Silver	<0.005	0.015
Zinc	0.003	0.22
Lead	0.0035	0.110
Chloride	10.75	524
Ammonia	<1.0	235
Bicarbonate	139	805
Uranium (U_3O_8)	0.098	24.4
Radium-226	27[a]	371[a]
Total dissolved solids	793	1324

[a] Units are pCi/ℓ.

example, Table 3.61 compares concentrations of substances in the groundwater before mining with those after mining but before restoration. These data are from tests conducted for the Irigaray Project (Wy77) and indicate those substances whose groundwater concentrations may be elevated by in situ leaching.

Radon emission during the restoration process has not been considered (Wy77, NRC78b, Ka78b). Because essentially all Ra-226 remains in the ore zone (about 97.5 percent), it appears reasonable to expect Rn-222 emissions to continue during restoration. A leached-out sector of the model mine will contain 156 Ci of Ra-226 in an aquifer volume of 2.53×10^5 m^3 (1.26×10^6 $m^3 \div 5$). Although no measurements have been made, it would appear that the restoration wastewater will contain about the same Rn-222 concentration as the pregnant leachate during leaching, 6.18×10^5 pCi/ℓ (Section 3.5.3). Assuming a pumping rate of 2,000 ℓ/min, a maximum of 650 Ci of Rn-222 will be released during each year of restoration, resulting in a maximum total release of 5,200 Ci during the estimated 8-yr restoration.

Restoration is presently in the experimental stages. No commercial-sized facility has reached that phase of operation. Although restoration by flushing appears feasible, there have been problems when alkaline lixiviants were used, particularly those containing ammonium ions. Ammonium is the preferred cation because sodium causes the clays to swell and plug the formation, and calcium forms an insoluble sulfate that also decreases the permeability of the formation. However, ammonium ions adsorb tightly on to clays by replacing the calcium and magnesium atoms in the clays. Montmorillonite, prevalent in the Texas mining areas, has extensive surface areas that result in very large ion-exchange capacities. Once adsorbed, the ammonium ions desorb at a very slow rate and prolong the restoration. It has been reported that after sweeping a leached ore zone with 10 ore zone volumes of water, the ammonium concentration of the water was reduced to 15 to 25 mg/ℓ (Ka78b). This concentration of ammonium may not be significant, although, under aerobic conditions, ammonium ions can be oxidized to the more toxic nitrate. In a deep aquifer, this oxidation process is not likely to occur, and, because of the very low leachability of ammonium ions from clays, any ammonium retained after restoration will move to surrounding aquifers at a very slow rate.

Several ongoing research studies are trying to solve the ammonium problem (Ka78b). Potassium is being tested as a cation replacement for ammonium in hopes that its adsorption and swelling characteristics will be favorable. Sweep solutions enriched in calcium and magnesium are being tested to determine if they will facilitate the flushing of the ammonium ion by replacing it on the clays by ion-exchange.

Restoration of the aquifer after mining stops is in the research stage. The adequacy of the restoration process and the procedures required will depend on a number of factors: the lixiviant used, concentration of specific ions in the lixiviant, the physical character of the stratigraphic unit, and the geochemical nature of the ore deposit. Undoubtedly, research will improve the process in the next few years. If the criteria of the restoration process are met, it is unlikely that there will be any adverse environmental impact from a properly restored aquifer.

Generally, the goal of reclaiming the site surface is to return the area to a state similar to that which existed naturally before mining. This often means one suitable for livestock grazing and wildlife habitat. The following site reclamation actions have been proposed (Wy77):

(1) Remove all structures and exposed pipes and plug all wells with concrete.

(2) After all impounded liquids have completely evaporated, cover the remains with overburden to a depth [2 m has been suggested at the Irigaray site (Wy77)] that will support plant growth and suppress Rn-222 emissions or transport and deposit the remains in a mill tailings impoundment.

(3) Before backfilling, dispose of the solids containing sufficient radioactivity to warrant removal by one of the methods suggested in Section 3.5.1.

(4) Grade surfaces of the backfilled ponds and all other barren areas to create a suitable topography and then revegetate them.

(5) Irrigate and fertilize sites to develop adequate plant cover.

(6) Maintain fences to prevent grazing by livestock until stable vegetative cover becomes established.

(7) Monitor reclaimed sites for radiation, verification of vegetative cover, and the absence of adverse erosion.

(8) Sample monitoring wells one year after restoration to verify aquifer restoration.

3.6 Other Sources

3.6.1 Mineral Exploration

During early exploration, uranium was identified by its mineral color, i.e., pitchblende from the Central City District in Colorado and carnotite in the Uravan Mineral Belt in Utah and Colorado. It was usually mined in conjunction with other metals and minerals. Later, when portable radiation survey meters became available, a substantial portion of the uranium findings (generally outcrops) were made by non-geologic prospectors (UGS54). Current uranium exploration uses extensive geological studies to locate formations with a strong potential for uranium ore content. These formations are then explored and field surveyed to verify the presence of ore. Much of the current exploratory activity is directed at expanding known deposits and mining areas.

As the surface and near-surface uranium deposits are found, mined, and depleted, exploration for reserves must be conducted at greater depths. The deeper uranium deposits, however, offer few radiometric clues on the surface regarding their location. In these cases, geologic studies and field work postulate the existence of promising geological formations. Actual exploration must be done by drilling. Drilling is also used to extend and explore known uranium producing areas.

There are two categories of drilling: exploratory and developmental. Exploratory drilling is used to sample a promising formation to determine if uranium ore is present. The drilling is generally done on a grid with the drill holes spaced 60 m to 1.6 km or more apart. Development drilling, to define the size and uranium content of the ore body, occurs when ore is struck in an exploratory hole. The development hole spacing ranges from 8 m to 100 m, depending on the characteristics and depth of the ore body. Usually, the same drilling equipment is used for both the exploratory and development drilling.

Ordinarily, there are three vehicles in a drilling unit. One vehicle carries and operates the drill rig, the second carries the drill rods, and the third carries water. Although the drill rig is a well-engineered, compact design, its physical size is increasing to meet the demands of deeper drilling (Personal communication with G. C. Ritter, 1979, Bendix Field Engineering Corp., Grand Junction, CO).

Early drilling (1948-1956) was predominantly done with percussion drills. These drills could drill to depths of about 76 m using 2.8 cm diameter drill steel. The drill bit was cooled and cuttings were removed from the drill hole by forcing air down the center of the drill stem. The cuttings (chips, sands, and dusts) were carried up and out of the drill hole by the air stream with velocities of 914-1520 m per minute (Ni76). The chips and coarse sands collected near the bore hole while the fine sands drifted and deposited around the drill site. Dusts, however, were free to drift with the winds.

Rotary drilling, used for boring deep holes, generally has replaced percussion drilling. Drill stems of 7.3 cm diameter are used to bore holes to depths of about 1300 m. Stems with diameters of 11.4 cm and larger are used for drilling holes in excess of 1300 m. The rotary drill bits are cooled generally in the same manner as percussion drills. When groundwater is encountered, water is used as a drilling medium and for removing cuttings. The cuttings are removed from the drill hole in the form of a slurry or drilling mud. They are usually stored in basins, either fabricated or dug in the ground. If unavailable, water is hauled to the drill site by truck. The drilling muds and water are stored in portable tanks or an earth impoundment for recirculation. After the drilling is completed, very often the cuttings are scattered and the drilling mud left at the site. This practice has been discouraged over the past 10 years in the Uravan area (Personal communication with G.C. Ritter, 1979, Bendix Field Engineering Corp., Grand Junction, CO). In some cases, the cuttings are disposed of in a trench and covered up with earth. Drilling muds are also sometimes covered. In either case, containment of the drilling wastes does not appear to be a prevalent practice.

Development drilling is conducted if ore is struck in an exploratory hole. The offset distance (i.e, the distance between development drill holes) is dependent on the previous history of the ore body sizes in the area. Offsetting may occur as soon as ore is struck, or it may be delayed until the exploratory drilling is completed.

The ore body may be evaluated by bore hole logging or by examining and analyzing cores. Core drilling, if used, usually begins at the top of the ore horizon. Ore (cores and cuttings) removed from the bore hole are sometimes removed from the drill site. In cases where the ore is not removed from the drill site, it remains with the dry cuttings or in the drilling muds. The drill hole collar is sometimes plugged with 0.9 - 1.5 m of concrete after the bore hole has been evaluated. In some states, the drill hole must be plugged to seal off aquifers in order to minimize groundwater contamination.

3.6.1.1 Environmental Considerations

By 1977, the uranium industry had completed 101 x 10^6 meters of surface drilling, with an all-time yearly high of 12 x 10^6 meters (DOE79). From 1958-1977, about 821,900 surface holes were drilled, resulting in 87.8 x 10^6 meters of bore holes. No statistics are available on the number of holes drilled from 1948-1958, but the annual and cumulative meters drilled for that period is known (DOE79). In order to estimate the number of drill rig placements for that period, the total annual meters of drilling was divided by the annual average bore hole depth. The average depth per bore hole was estimated by plotting the average annual bore hole depths for 1958-1977 then using that data to estimate the annual bore hole depths for 1948-1957 by linear regression analysis (Fig. 3.22).

The data points in Fig. 3.22 appear to fall into two groups: 1958-1966 and 1966-1977. The average drilling depth of the 1966-1977 group of data points probably reflects the deep drilling in the Grants, New Mexico area that became significant in 1969. Using this information, the 1948-1958 average drilling depths were estimated from regression analysis using the 1958-1966 data points only. Table 3.62 is a summary of the DOE drilling data and the number of estimated bore holes by type and year.

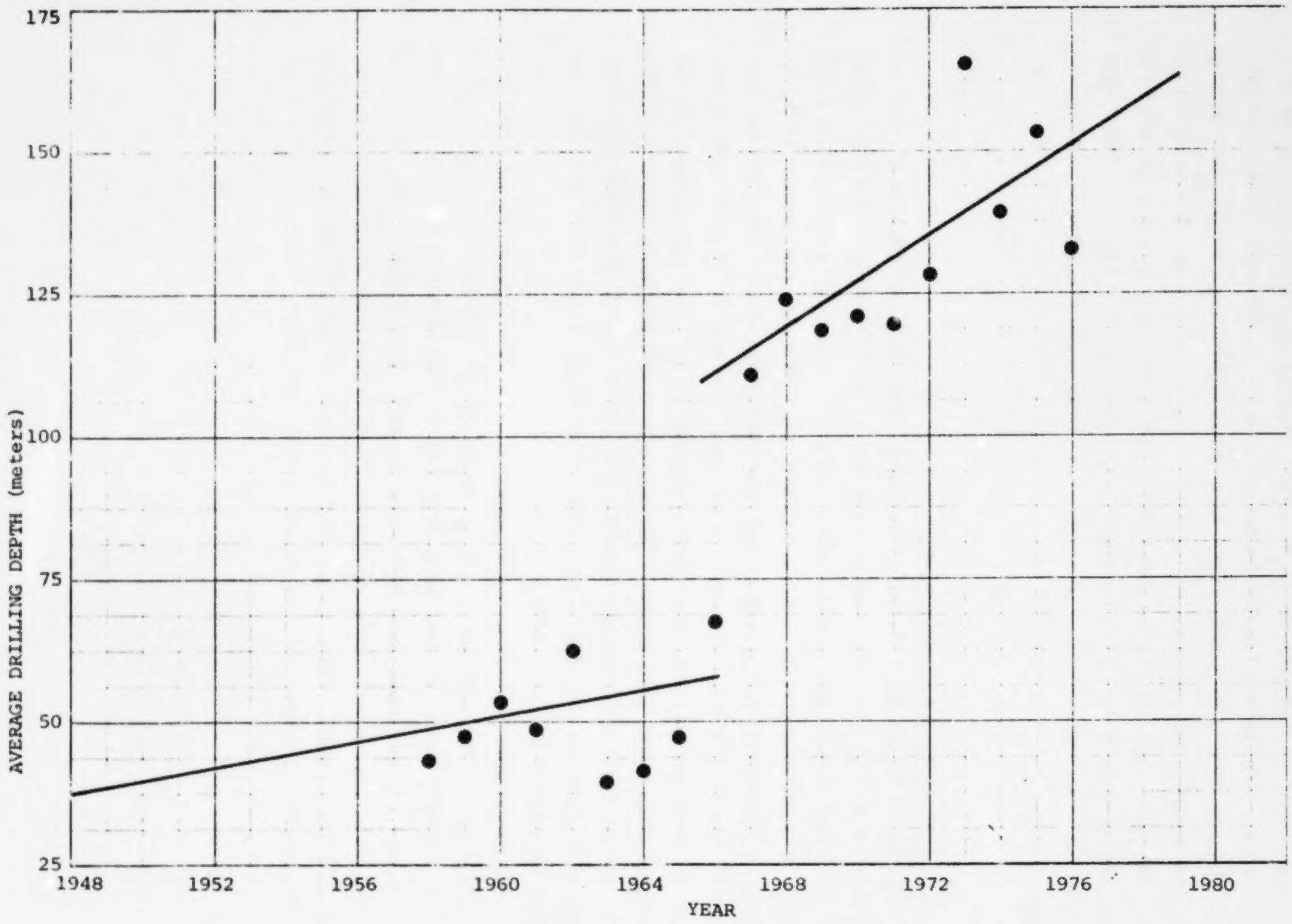

Figure 3.22 Average depth of exploratory drilling in the U.S. uranium industry from 1948 to present.

Table 3.62 Estimates of exploratory and development drill holes (1948-1979)

Year	Surface Drilling (10^6 Meters)		Average Hole	Number of Holes	
	Exploration	Development	Depth(Meters)	Exploration	Development
1948	0.052	0.012	38.1(a)	1,360(b)	320(b)
1949	0.110	0.016	38.1(a)	2,880(b)	424(b)
1950	0.174	0.063	39.6(a)	4,380(b)	1,600(b)
1951	0.329	0.106	41.1(a)	8,000(b)	2,580(b)
1952	0.415	0.091	41.1(a)	10,100(b)	2,220(b)
1953	1.11	0.112	42.7(a)	26,100(b)	2,620(b)
1954	1.24	0.169	42.7(a)	29,000(b)	3,950(b)
1955	1.61	0.232	44.2(a)	36,300(b)	5,260(b)
1956	2.22	0.457	45.7(a)	48,600(b)	10,000(b)
1957	2.24	0.564	45.7(a)	49,000(b)	12,300(b)
1958	1.15	1.06	45.7	25,300	22,900
1959	0.722	1.00	48.2	16,300	19,600
1960	0.427	1.28	53.9	7,340	24,400
1961	0.402	0.972	50.0	8,260	19,300
1962	0.451	0.741	61.9	6,440	12,900
1963	0.268	0.604	39.6	8,470	13,500
1964	0.294	0.381	42.4	5,970	9,910
1965	0.354	0.289	47.5	6,230	7,330
1966	0.549	0.731	67.7	5,750	13,200
1967	1.67	1.62	110	12,800	16,900
1968	4.97	2.30	125	38,500	19,500
1969	6.25	2.86	120	47,900	28,000
1970	5.49	1.69	122	44,000	14,900
1971	3.47	1.23	121	28,400	10,400
1972	3.60	1.10	128	26,900	9,710
1973	3.29	1.70	146	22,600	11,700
1974	4.88	1.83	168	27,400	12,300
1975	5.03	2.74	139	34,300	21,600
1976	5.94	4.48	154	40,400	27,200
1977	7.89	4.45	132	62,600	30,900
1978	10.8	5.24	155(a)	69,200(b)	33,700(b)
1979	9.94	5.18	158(a)	62,700(b)	32,700(b)
TOTAL	286	149		823,000	454,000

(a) Indicates estimated average depth from Fig. 3.22.

(b) Indicates number of drill holes estimated by dividing the annual exploration and surface drilling depths by the average hole depth.

Cuttings produced by drilling can degrade the drill site area and the local air quality. For convenience of evaluation, the cuttings are divided into two general categories--dusts and wastes. The dusts are drilling fines that become airborne, and wastes are drilling chips and sands deposited around the borehole. The maximum dust production occurs when compressed air is used solely for cleaning the boreholes. Generally the drilling industry uses foaming agents injected into the compressed air stream to help remove drill cuttings. The foam traps and contains the fine particulates and substantially reduces the airborne dust. In practice, the drillers minimize airborne dust, because it causes excessive wear on engines and compressors. Dust production also indicates improper drilling energy being used to grind up cuttings in the borehole rather than bore. Occasionally some water may also be injected into the air stream to remove cuttings and to keep the drill hole from collapsing when loose materials are encountered.

There are some estimates of airborne dust production and general assumptions concerning drilling practices (Private communication with Mr. T. Price, Bendix Corp., Grand Junction, CO and E. Borgerding, Borgerding Drilling Co. Inc., Montrose, CO). They are as follows:

(1) The ratio by weight of the chips, sands, and dusts produced by drilling is approximately 60:37:3, respectively (i.e., 3 Kg of every 100 Kg of cuttings removed from a borehole is available as airborne dust).

(2) Fifty percent of all drill holes are wet (mud) drilled and 50 percent are air drilled; ninety-five percent of the latter are drilled using mist or foam (i.e., 2.5 percent are dry-drilled).

(3) The first 6.6 m of all drill holes are drilled dry (i.e., no mist or foam is used).

We estimated dust production from contemporary drilling by averaging drilling data from Table 3.62 for the years 1975 through 1979. The average depth of the holes for this period is 148 m. The annual average numbers of exploration and development holes are 53,800 and 29,200, respectively. Airborne dust production from those holes that are drilled with mud (wet), foam, or mists (97.5 percent of both the exploratory and development holes) will originate only from the first 6.6 m depth. The weight of dust generated per hole will be as follows:

Airborne dust (kg) = Volume of borehole (m^3) x density (kg/m^3) x airborne dust fraction (.03) per drill hole

$= (\pi r^2 h)\,(2000\frac{\text{kg}}{\text{m}^3})(0.03)$ where $h = 6.6$ m

$r = 0.0865$ m (assumed average radius of 2 bit sizes $r = 7.3$ cm and 10 cm) (Pe79)

$$= (3.14)(7.48 \times 10^{-3})\ \text{m}^2 \times 6.6\ \text{m} \times 2000\ \frac{\text{kg}}{\text{m}^3} \times 0.03$$

$$= 9.3\ \text{kg}$$

The average weight of airborne dust (kg) produced from all contemporary annual drilling (first 6.6 m) is

$$83{,}000\ \text{drill holes} \times \frac{9.3\ \text{kg}}{\text{drill hole}} = 7.7 \times 10^5\ \text{kg}.$$

The annual total weight (kg) of airborne dust produced from 2.5 percent of the annual number of drill holes bored (dry) where no mud, mists, or foams are used

$$= 83{,}000\ \text{drill holes} \times \frac{148\ \text{m}}{\text{drill hole}} \times 0.025 \times 47\ \frac{\text{kg}}{\text{m}}\ \text{cuttings} \times$$

$$0.03\ \text{kg dust/kg cuttings} = 4.3 \times 10^5\ \text{kg/yr}. \qquad (3.13)$$

The total weight of airborne dust produced annually from each dry-drilled borehole is 209 kg.

Assuming that each development hole penetrates the 3.6 m ore body, the total amount of airborne ore and sub-ore dust produced from development drilling annually is

$$29{,}200\ \frac{\text{drill holes}}{\text{yr}} \times \frac{3.6\ \text{m (ore and sub-ore)}}{\text{drill hole}} \times \frac{47\ \text{kg cuttings}}{\text{m}} \times$$

$$0.03\ \text{kg dust/kg cutting} \times 0.025 = 3.7 \times 10^3\ \text{kg}. \qquad (3.14)$$

The total weight of airborne ore and sub-ore dust produced from each development drill hole (no mud, mists, or foams used) is 5.1 kg.

The estimated annual quantity of ore and sub-ore brought to the surface by contemporary drilling equals:

$$29{,}200\ \frac{\text{drill holes}}{\text{yr}} \times \frac{3.6\ \text{m}}{\text{drill hole}} \times \frac{47\ \text{kg cuttings}}{\text{m}} \qquad (3.15)$$

$$= 4.9 \times 10^6\ \frac{\text{kg}}{\text{yr}}\ \text{or}\ 4.9 \times 10^3\ \frac{\text{MT}}{\text{yr}}.$$

Most of the ore will remain at the drill site with drilling muds or with the drilling wastes around the drill holes. Since the ore most usually will be the last material removed from the boreholes, it will be deposited on the surface of the cuttings and drilling muds. This will expose the ore to the elements and subject it to erosion.

3.6.1.2 Radon Losses from Drill Holes

When the development drill penetrates an ore body, some of the ore and sub-ore bearing formations will be exposed to air in the drill hole. Some of the radon gas produced in the ore can enter into the air in the drill hole and escape to the atmosphere. The mechanisms affecting the release rate of radon from boreholes are poorly understood. Tanner observed a wide variation in radon concentrations as a function of depth in an open borehole as compared to a closed borehole (Ta58). Tanner also noted that strong winds could significantly reduce the total radon content of an uncovered borehole. Since so little is known about radon discharges from development boreholes, radon losses in this report are assessed on a "worst case" basis using the following assumptions:

1. The drill hole is not plugged.
2. About 3.6 m of ore and sub-ore were drilled.
3. All radon released into the borehole escapes to the atmosphere.
4. The average grade of the ore and sub-ore is 0.17 percent.
5. No water accumulates in the borehole.

The surface area of the borehole passing through the ore and sub-ore body is

$$2\ \pi rh = 2 \times 3.14 \times 0.0865\ \text{m} \times 3.6\ \text{m} = 2.0\ \text{m}^2. \qquad (3.16)$$

The radon release rate is estimated for ore and sub-ore in the borehole using an exhalation rate of 0.092 Ci/m^2 per year per percent of U_3O_8 (Ni79). The quantity of radon (Q) per development hole escaping per unit time is

$$\frac{0.092\ \text{Ci}}{\text{m}^2\ \text{yr}\ \%} \times 0.17\% \times 2.0\ \text{m}^2 \times \frac{1}{3.15 \times 10^7\ \text{sec/yr}} \times 10^{12}\ \frac{\text{pCi}}{\text{Ci}} = 990\ \text{pCi/sec} \qquad (3.17)$$

The total quantity of radon per annum escaping from all development holes drilled through 1979

$$= 4.5 \times 10^5\ \text{drill holes} \times \frac{990\ \text{pCi}}{\text{sec-drill hole}} \times 3.15 \times 10^7\ \text{sec/yr}$$

$$\times \frac{1}{10^{12}\frac{\text{pCi}}{\text{Ci}}}$$

$$= 14{,}000\ \text{Ci/yr} \qquad (3.18)$$

The "worst case" estimate can be modified by assuming 50 percent of the holes are wet and 30 percent of the remaining holes are plugged or have collapsed. In this case, the total source term would be about 4,900 Ci/yr. Since about 31 percent of the development drill holes are at surface mines and are consumed by the pits, the annual Rn-222 release from the remaining holes will be 3,400 Ci/yr.

3.6.1.3 Groundwater

Progressively deeper holes are being drilled as the ore bodies near the surface become depleted. As the drilling depths increase, one or more aquifers may be intercepted by a drill hole, and an aquifer with poor water quality may be connected with an aquifer with good water quality. Depending on the direction of flow, the quality of water may be downgraded in a good aquifer. Most states require some plugging of the drill holes to seal the aquifer in order to maintain water quality. Adequate plugging of the drill holes requires a conscientious effort on the part of the driller and the regulatory agency. Since the movement of groundwater is relatively slow, the change in the quality of water in an aquifer will not be apparent for some time. Thus, it may take a long time to correct the quality of water in a downgraded aquifer.

3.6.1.4 Fumes

It is estimated (Pe79) that 11.2 liters of diesel fuel are needed to drill 1.0 m. In 1979, the average borehole depth was estimated to be 158 m and would require about 1770 liters of diesel fuel. This fuel would be burned at a rate of approximately 173 liters per hour. Some individual holes, however, are drilled in excess of 914 m and require 10,200 liters of diesel fuel. It is estimated that about 170 million liters of diesel fuel were consumed for all 1979 drilling.

The principal emissions from the drilling power sources are particulates: sulfur oxides, carbon monoxide, nitrogen oxides, and hydrocarbons. Because of the transient nature of the drilling, these releases are not expected to substantially lower air quality over time.

3.6.1.5 Model Drilling

About 1.3×10^6 holes have been drilled and bored for all uranium mining from 1948 through 1979 for approximately 3000 mines. This would amount to about 430 holes per mine. Thirty-six percent of the holes were for development drilling, and 64 percent were for exploratory drilling. Assuming that 50 percent of the exploratory and development holes are air drilled (see Section 3.6.1.1), the airborne dust production for an average mine may be estimated as follows:

Airborne dust from all drill holes (first 6.6 m of depth air drilled dry)

$$= 430 \text{ drill holes} \times \frac{9.3 \text{ kg}}{\text{drill hole}} = 4000 \text{ kg.} \qquad (3.19)$$

Airborne dust from all dry air drilling, less the first 6.6 m, (3.20)

$$= \left(430 \text{ drill holes} \times \frac{209 \text{ kg dust}}{\text{drill hole}} \times 0.5 \times 0.05\right) - 100 \text{ kg} = 2100 \text{ kg.}$$

Airborne ore and sub-ore dust produced by dry air drilling

$$= 430 \text{ drill holes} \times 0.36 \times 0.5 \times 0.05 \times \frac{5.1 \text{ kg dust}}{\text{drill hole}} = 20 \text{ kg.} \qquad (3.21)$$

Total airborne dust produced from all drilling at an average mine site

$$= 4000 \text{ kg} + 2100 \text{ kg} = 6100 \text{ Kg} = 6.1 \text{ MT.} \qquad (3.22)$$

Twenty kilograms of the total dust produced will be ore and sub-ore dusts. The Rn-222 emissions from the bore holes at an average mine site would be

$$430 \text{ drill holes } (0.5)(0.36) \left(\frac{990 \text{ pCi}}{\text{sec-drill hole}}\right) = 7.7 \times 10^4 \frac{\text{pCi}}{\text{sec}}, \qquad (3.23)$$

or 2.4 Ci/yr.

Development drill holes at a surface mine would be consumed by the pit.

Tables 3.63--3.65 show airborne particulate source terms for uranium drilling for individual drill holes and for an average uranium mine. Table 3.63 lists the airborne dust produced for each type exploratory and development borehole; Table 3.64 summarizes the quantity of airborne dust produced by all types of drilling at an average mine site; and Table 3.65 lists the pollutants emitted from a drill rig power source.

3.6.2 Precipitation Runoff from Uranium Mines

Unquestionably, overland flow or surface runoff from precipitation transports dissolved and suspended contaminants from mining areas to the offsite environment. Unfortunately, the significance of this pathway rela-

Table 3.63 Estimated source terms per borehole for contemporary surface drilling for uranium

Type of Drilling	Thickness of Ore and Sub-ore Bodies (m)	Airborne Dust Production		Airborne Ore and Sub-Ore Dust Production	
		Total(kg)	Rate(kg/min)[a]	Total(kg)	Rate(kg/min)[a]
Exploratory					
Air (dry)	NA[b]	209	0.27	NA	NA
Air (mist or foam)	NA	9.3	0.27	NA	NA
Wet (mud)	NA	9.3	0.27	NA	NA
Development					
Air (dry)	3.6	209	0.27	5.1	0.27
Air (mist or foam)	3.6	9.3	0.27	NA	NA
Wet (mud)	3.6	9.3	0.27	NA	NA

[a] Based on an air drilling rate of 11.5 m/hr.

[b] NA - not applicable.

Table 3.64 Airborne dusts produced at an average mine site from exploratory and development drilling

Type of Drilling		Quantity of Airborne Dust (kg)
All types (first 6.6 m depth)		4,000
Air drilling (dry)		2,100
	Total	6,100 kg[a]

[a] Twenty kg of the total will be ore and sub-ore dusts.

Table 3.65 Estimates of emissions from drill rig diesel power source

Pollutant	Production Rate ($kg/10^3$ liters fuel)	Quantity[a] (kg/drill hole)	Rate[a] (kg/hr)
Carbon monoxide	12.2	20.2	1.5
Hydrocarbons	4.49	7.4	0.55
Nitrogen oxides	56.2	93	6.9
Aldehydes	0.84	1.39	0.10
Sulfur oxides	3.74	6.2	0.46
Particulates	4.01	6.6	0.49

[a] Based on a drilling rate of 11m/hr.

Source: EPA77b.

tive to uranium mines is highly site specific and poorly understood. Very few field studies of runoff from uranium mining areas have been conducted, and what field data do exist frequently relate to the combined and probably greater influences of mine water discharge and milling. Most of the NRC regulations apply to mill operations, since mining is generally exempt from the agency's charter. The EPA regulations (Environmental Radiation Protection Standards for Nuclear Power Operations; 40 CFR Part 190) applicable to the uranium fuel cycle establish dose limits for individuals to provide protection for populations living in the vicinity of uranium mills. Uranium mines are excluded, and so are liquid effluent guidelines for ore mining and dressing (40 CFR 440, Subpart E). Regulations being developed under the Resource Conservation and Recovery Act (RCRA) of 1976 apply to radioactive wastes not covered by the Atomic Energy Act of 1954, as amended. Solid and liquid waste categories will be defined in forthcoming EPA regulations developed under RCRA, but it is not anticipated that runoff from mined lands will meet the waste characteristics in the regulations. Similarly, the Federal Water Pollution Control Act Amendment of 1972, the Clean Water Act of 1977, the Safe Drinking Water Act, and State regulations in general do not address surface runoff effects of mining. Without the regulatory base, studies and field data are, not surprisingly, rather scarce. In New Mexico, the State's 208 Water Quality Management Plan calls for, among other things, improved data collection on runoff from active and inactive tailings piles and from drilling, exploration, and development activities such as access road and drill site construction (So79).

We have not estimated chemical transport by overland flow because of the limited time for the study. But, it is reasonable to expect that such transport may be quite significant in an arid and semiarid climate where much of the precipitation that does infiltrate is discharged back into the atmosphere as water vapor. This has been well demonstrated in the case of uranium mill tailings (K178). Water moving back out of the soil transports dissolved salts that are deposited on the soil surface when the carrier (water) evaporates. Subsequent precipitation further transports these salts downward into the soil and laterally to offsite areas. So-called "blooms" of salt crystals, composed mainly of sulfate and chloride compounds, characterize uranium ore bodies, mill tailings piles, and mine wastes in a number of Western States, and we must presume that such salts solubilize in runoff.

This also indicates that there may be large concentrations of contaminants available for plant uptake. Molybdenum, in particular, is one of the toxic elements on such blooms, and uranium is also highly suspect. Selenium, arsenic, and vanadium may also be present, since their anions are mobile under oxidizing conditions characteristic of the near-surface, unsaturated zone (Fu78).

Overburden has been used extensively to backfill surface mines operating since the early to mid 1970's, but this is not true at many if not most older and now inactive mines. Erosion of these piles by water and wind may present the greatest problem (Ka75). Using overburden to construct access roads and dikes distributes contaminants in the local environment and may aggravate air and water pollution. Considering that 75 percent of the overburden has a grain size exceeding 2000 μm (see Table 3.12), it is unlikely that widespread physical transport will result from overburden piles. However, using overburden for roads decreases the grain size. The association of uranium and progeny with the smaller sediment-size fractions, by a factor of 2.5, increases the potential for transport by overland flow.

Tables 3.15, 3.16, and 3.19 show stable and radioactive trace elements in ores, sub-ore, and overburden from uranium mines. Understandably, uranium, thorium, and radium are high. Arsenic, selenium, vanadium, and molybdenum are almost always closely associated with uranium. Barium, zinc, manganese, copper, iron, and potassium may also be associated in certain mineral provinces and districts. Mercury and cadmium are occasionally present (Th78). There is no consistent relationship between ore grade and trace metal content in selected New Mexico and Wyoming study areas (Wo79).

Particularly in the case of active or recently active mines, surface runoff is collected with dikes and ditches that route water to settling ponds. Water spray or chemical additives can control road dust. They are commonly used in the active mining stage, but almost never used during exploratory drilling. Grading piles to a slope of 3:1 or less also helps to reduce runoff (St78), and this practice is becoming common in Texas and Wyoming. Proper planting techniques further reduce runoff by increasing infiltration and decreasing sediment transport.

The significance of surface runoff from mining areas as a dispersal mechanism was investigated as part of this study (Wo79) (see also Section 3.2.3.2). We examined stable and radioactive trace elements in soils affected by runoff from ore, sub-ore, and mine waste/overburden piles from one active surface mining area in Wyoming and two inactive areas (surface and underground mines) in New Mexico. Although there was evidence of offsite movement of uranium and radium at all sites, transport is limited and decreases with distance from the site. In Wyoming, pollutant releases from the mine studied do not reach nearby water courses although onsite transport of stockpiled ore as a result of precipitation runoff does occur.

A U.S. Bureau of Mines (BOM, no date) study of strip and surface mining operations and their effects in the United States involved questionnaires, literature survey, and onsite examinations of 693 selected sites, among which were uranium mines in New Mexico and Wyoming. At 60 percent of the sites, on a national basis, there were no serious problems because vegetation was reestablished and the slope of the land was gentle both before and after mining. Thirty percent of the sites had eroded to depths of 0.3 m or less, and the remainder were gullied to greater depths. There were sediments from mined lands in 56 percent of the ponds and 52 percent of the streams on or adjacent to the sample sites. Spoil bank materials ranged in pH from 3 to 5 at 47 percent of the sites and are thus not amenable to plant growth. Field observations substantiate that mined land areas, be they former forests or grasslands, did not return to the pre-mining condition. Idle land increased almost fourfold because of mining. The study concluded that natural processes need to be strongly supplemented if mined sites are to revert to former uses. Since only 6.3 percent of lands mined for uranium were reclaimed from 1930 through 1971 (Pa74), it seems reasonable to conclude that there are increased sediment loads, gullying, and poor revegetation at most older inactive mines that were poorly stabilized, if at all.

The Bureau of Mines study concluded that peak sediment loads in runoff are characteristic of areas with high intensity storms and steep slopes, particularly during and shortly after mining. Such problems are less severe in arid regions, but large quantities of sediment are discharged from mine workings, spoil heaps, and access roads. In some instances, effects of wind and water erosion on steep spoil banks in arid lands are evident many years after abandonment. In areas outside Appalachia, 86 percent of the areas investigated had sufficient runoff control, and those areas where there was a problem almost exclusively involved coal, phosphate, manganese, clay, and gold.

Incidences of radioactive contamination of local surface water have been documented for the Shirley Basin uranium mine (Utah International, Inc.) in Wyoming (Ha78). The most pronounced changes in water and stream sediment quality coincided with initial strip mining and mill processing operations. Early acid-leach solution mining also had a decided impact. Pollutant loadings from overland flow, per se, were not determined but are presumed to be minor compared to aqueous discharges from mines and mills. These findings contradict those of an earlier study (Wh76) of the same mine. Soil and vegetation collected from 1971 through 1975 at 28 stations in the vicinity of the mine were analyzed for gross alpha and gross beta (1971 to 1974) and total uranium, Ra-226 and Pb-210 (1975). The study (Wh76) concluded that--

1. concentrations of the foregoing parameters were extremely variable but reasonably consistent with previously reported information;
2. there is no evidence that radionuclide concentration in soil or vegetation collected from routine monitoring stations are changing with time;
3. concentrations of radioactivity in soil and vegetation correlate with distance from the mill area to a distance of 1.2 miles; and
4. measurable ecological effects from radiation in the environs of the Shirley Basin mine cannot be demonstrated.

The absence of statistically significant soil and vegetation contamination from the mine versus the mill is noteworthy. Overall, vegetation tends toward higher alpha and beta concentrations than soil, except at the close-in, upwind sampling areas. This selective concentration in vegetation suggests aerial deposition of contaminated dust particles on vegetation, with some additional possibility for root uptake.

Estimates of surface drilling for uranium reveal that relatively large land areas are involved. The volume of cuttings removed from borings in the period 1948 through 1979 is calculated using 286 x 10^6 m of exploratory drilling and 104 x 10^6 m of development drilling (from Table 3.62). We assumed that 30 percent of the mines are surface mines, which eliminates the borings and related debris. Thus the value of 149 x 10^6 (in Table 3.62) is reduced by 30 percent. Average diameter for 8.5 x 10^6 m of borings in the period 1948 through 1956 is 2.8 cm versus 7.3 cm for the period 1957 through 1979 (see Section 3.6.1) when 426 x 10^6 m of drilling took place. A sample calculation for the volume removed from borings made in the period 1975-1979 follows:

$$\begin{aligned} V &= \pi r^2 h \\ &= (3.14)\left(\frac{7.43\ \text{cm}}{2}\right)^2 (146\text{m}) \\ &= 0.632\ \text{m}^3 \end{aligned} \tag{3.24}$$

Assuming a bulk density of 2000 Kg per m^3, each boring results in 1265 Kg of cuttings at land surface. There were 415,300 borings, resulting in 263,000 m^3 of cuttings. Assuming that the average thickness of cuttings is 0.5 m, 526,000 m^2 or 0.53 Km^2 is affected. The inclusive area affected by drilling from 1948 through 1979 is 3.6 Km^2.

Table 3.66 summarizes the surface areas affected by mine wastes, ore piles, and exploration and development activities. Maximum use was made of data developed elsewhere in this report on the number of mines, waste pile dimensions and surface areas, and the summary of exploration and development. The estimate is, at best, a first approximation and needs considerable refinement.

For example, grain size, degree of consolidation, slope, vegetative cover, and other characteristics may vary considerably between ambient soil and rock materials versus mine wastes. The latter very often occur in steep, unvegetated piles and are composed of easily-eroded, friable sandstone, boulders, and fines. It is likely, therefore, that the sediment yield on a mass per time per area basis exceeds that of the surrounding areas; thus the estimate developed below may well be on the low side.

Sediment yields from areas affected by various mining operations are roughly estimated from consideration of land areas affected and unit soil loss values for the surrounding regions. Actual values for individual tailings or waste piles may be considerably different, but refining the values given will require additional analysis beyond the scope of the present study.

Potential coal mining lands in the Northeastern Wyoming range lose soil at rates of 4.8 to 167 $m^3/Km^2/yr$ (Ke76). Upland erosion and stream channel erosion in the Gillette study area are not generally serious problems, since land dissection is presently minimal and vegetative cover is well established. The potential for increased sediment yield is large, if vegetative cover were to be reduced or eliminated and slopes steepened because of mining. Certainly, during active mining, these conditions will be at least locally present. Erosion rates of 600 to 1,100 $m^3/Km^2/yr$ from mined lands in the South Powder River Basin are expected, and they are reasonably close to

Table 3.66 Sediment yields in overland flow from uranium mining areas

Source Term	Factor	No. Installations	Cumulative Source, Km^2	Annual Sediment Loading, m$^{3(a)}$
Active Mines				
Underground				
Ore piles	603 m^2/mine	251 mines	0.15	143
Sub-ore piles	26,700 m^2/mine	251 mines	6.7	6385
Waste rock piles	26,700 m^2/mine	251 mines	6.7	6385
Surface				
Ore piles	4.15 x 10^3 m^2/mine	36 mines	0.15	143
Sub-ore piles	67 x 10^3 m^2/mine	36 mines	2.4	2287
Overburden piles	380 x 10^3 m^2/mine	36 mines	13.7	13056
Inactive Mines				
Underground				
Waste piles and sub-ore	4.07 x 10^3 m^2/mine	2108 mines	0.86	820
Surface				
Overburden and sub-ore	6.73 x 10^4 m^2/mine	944 mines	64	61000

Table 3.66 (Continued)

Source Term	Factor	No. Installations	Cumulative Source, Km^2	Annual Sediment Loading, m$^{3(a)}$
Exploration and Development				
Drilling				
1948-1979	435 x 10^6 m	1.28 x 10^6 borings	3.6	3431
1975-1979	1265 kg/boring	415,300	0.53	506
Access roads and pads	1.25 acres or 0.5 ha/boring	1.28 x 10^6	6500	6.2 x 10^6

(a) Assumes average sediment yield of 953 m^3/Km^2.

Note.--Data in this table are based on average mine vs. average large mine as defined in Section 3 of report.

natural, pre-mining conditions (R. Loeper, Soil Conservation Service, 1979, personal communication). At the Bear Creek mine, the reclamation design calls for maximum losses from overburden piles of 1,100 $m^3/Km^2/yr$ initially and 600 $m^3/Km^2/yr$ after the first 3 years. In general, erosion and soil loss from uranium mining in this part of Wyoming is not a significant problem, mainly because of reclamation by industry. Sediment yields in the Grants Mineral Belt range from 95 to 240 $m^3/Km^2/yr$ in the area of the large Jackpile-Paguate surface mine to 500 to 1,400 $m^3/Km^2/yr$ near the underground mining centers around Smith Lake, Ambrosia Lake, and Churchrock (P. Boden, Soil Conservation Service, 1979, personal communication). Considering both the Wyoming and New Mexico model mine areas, this study used an overall average annual soil loss rate of 953 m^3/Km^2. This average sediment yield rate is based on studies by the Soil Conservation Service of large areas in New Mexico, Wyoming, and other Western States.

In summary, the total land area directly affected by uranium mining is about 6600 Km^2. Assuming an overall average sediment yield of 953 m^3/Km^2, annual sediment transported by overland flow is approximately 6.3×10^6 m^3. Obviously exploration and development activities affect the greatest area (6500 Km^2), but they do not necessarily have the greatest impact. Exploration and development, for example, affect large areas, but most of the area affected is a result of constructing access roads and drill pads. Whereas sediment yields from ore, sub-ore, overburden, and waste rock is estimated at 90,000 m^3 per year. Surface mining, although it supplies only about 30 percent of U.S. production, affects the second greatest area (80 Km^2). We have not attempted to characterize the quality of sediment runoff. The fate of these sediments is very poorly understood and has not been the subject of intensive investigation. Further study in the area of intensive surface mining such as in Texas and Wyoming is needed to determine changes in erosion rates resulting from mining and to quantify the contaminant flux and fate.

3.7 Inactive Mines

3.7.1 Inactive Surface Mines

For generic purposes, a model inactive open pit or surface uranium mine must be defined in order to estimate the environmental impact from this type of mining. We have assumed that an inactive surface mine has a single hole or pit in the ground, with all of the materials (wastes) stacked into piles adjacent to the pit area. The size or volume of the pit would be approximately equal to the volume of the ore and wastes removed from it. Since only 6.3 percent of all of the land used for uranium mining has been reclaimed from 1930 through 1971 (Pa74), no credit for reclamation is given to the model mine.

Ideally, the model mine size could be established by averaging the ore and waste production for each inactive surface mine. Unfortunately, these statistics are either not thoroughly documented or they are retained as company confidential information. In lieu of specific information, the model surface mine size was established from annual ore and waste production statistics for all surface mines, divided by the number of inactive surface mines.

Table 3.67 is a summary of inactive mines, obtained from the Department of Energy mine listing. The mines are listed by type, surface and underground. Most of the inactive surface mines are in Colorado, Utah, Arizona and New Mexico. For model derivation purposes, we assumed that there are presently 1250 inactive surface uranium mines.

Table 3.68 lists mine waste and ore production information from 1932 to 1977. Uranium mine waste and ore production statistics, on an annual basis, were available from both surface and underground uranium producers from 1959 to 1976 (DOI59-76). Annual uranium ore production statistics for each uranium mining type (surface and underground) are available for 1948 to 1959 (DOE79) and for combined uranium production from 1932 to 1942 (DOI32-42). In order to estimate waste production for the years prior to 1956, the annual mine type ore production records were multiplied by waste-to-ore ratios. These ratios were estimated from published 1959 to 1976 ore and waste production statistics (DOI59-76). Very little uranium ore was mined from 1942 to 1948, since most of the uranium was obtained by reprocessing vanadium and radium tailings (personal communication with G. Ritter, Bendix Field Engineering Corp., Grand Junction, CO, 1979). The annual waste production for surface mining from 1948 to 1959 was estimated by extrapolating known waste-to-ore ratios (1959 to 1976) through the 1948 to 1959 time period using a "best fit" regression analysis (Fig. 3.23). This method cannot be used to estimate waste-to-ore ratios because the waste production is finite and will always occur, and also surface mining for uranium essentially began in 1950.

Table 3.67 Consolidated list of inactive uranium producers by State and type of mining

State	Surface	Underground	Percent of Total Surface Mines	Percent of Total Underground Mines
AL	0	1	0.0	< 0.1
AZ	135	189	11	9.3
CA	13	10	1.0	0.49
CO	263	902	21	44
ID	2	4	0.16	0.20
MT	9	9	0.72	0.44
NV	9	12	0.72	0.59
NJ	0	1	0.0	< 0.1
NM	34	142	2.7	7.0
ND	13	0	1.0	0.0
OK	3	0	0.24	0.0
OR	2	1	0.16	< 0.1
SD	111	30	8.9	1.5
TX	38	0	3.0	0.0
UT	378	698	30	34
WA	13	0	1.0	0.0
WY	223	32	18	1.6
Total	1246	2031		

Table 3.68 Uranium mine waste and ore production (MT x 1000)

Year	Surface Mining		Underground Mining		Surface Mining Waste/Ore	Underground Mining Waste/Ore	Total Ore Produced By Surface and/or Underground Mines
	Crude Ore	Waste	Crude Ore	Waste			
1977	5059	237800	4305	3487	47	0.81	9364
1976	4238	190700	3569	2605	45	0.73	7807
1975	3809	139700	2485	2195	37	0.88	6295
1974	3510	129700	2222	1424	37	0.64	5732
1973	3800	182300	1614	934	48	0.58	5414
1972	3447	155100	2439	593	45	0.24	5886
1971	2656	120200	2836	858	45	0.30	5492
1970	2490	76870	3304	962	31	0.29	5794
1969	1653	81000	3171	1184	49	0.37	4824
1968	1989	31360	3382	1163	16	0.34	5371
1967	1393	32510	2897	1024	23	0.35	4290
1966	905	24400	2777	863	27	0.31	1768
1965	1630	17710	3055	809	11	0.26	4685
1964	2344	26680	3227	941	11	0.29	5571
1963	3578	33120	3575	946	9.0	0.26	7153
1962	2895	44640	4892	1087	15	0.22	7787
1961	3051	42500	5017	1117	14	0.22	8068
1960	2691	73570	5104	1868	27	0.37	7795
1959	2494	46790	3796	941	19	0.25	6290
1958	2139	19240	2558	690	9.0[a]	0.27[b]	4697
1957	1462	11700	1888	510	8.0	0.27	3350
1956	1131	9048	1595	414	8.0	0.26	2726
1955	339	2650	1043	271	8.0	0.26	1382
1954	241	1930	762	198	8.0	0.26	1003

Table 3.68 (continued)

Year	Surface Mining		Underground Mining		Surface Mining Waste/Ore	Underground Mining Waste/Ore	Total Ore Produced By Surface and/or Underground Mines
	Crude Ore	Waste	Crude Ore	Waste			
1953	162	1300	503	126	8.0	0.25	665
1952	59	472	341	85	8.0	0.25	400
1951	25	203	289	73	8.0	0.25	314
1950	21	167	207	50	8.0	0.24	228
1949			156	37		0.24	156
1948			34	8.3		0.24	34
1947			0			0.24	
1946			0			0.23	
1945			0			0.23	
1944			0			0.23	
1943			0			0.22	
1942			0			0.22	
1941			0.824	0.181		0.22	0.824
1940			0.7221	0.151		0.21	0.722
1939			5.68	1.19		0.21	5.68
1938			3.89	0.817		0.21	3.89
1937			1.55	0.310		0.20	1.55
1936			1.31	0.261		0.20	1.31
1935			1.03	0.207		0.20	1.03
1934			0.230	0.0461		0.20	0.230
1933			0.047	0.00896		0.19	0.047
1932			0.0553	0.0105		0.19	0.0553

[a] Waste to ore ratios from 1950 - 1958 estimated from 1959 - 1972 ratios.

[b] Waste to ore ratios from 1932 - 1958 estimated from 1959 - 1972 ratios.

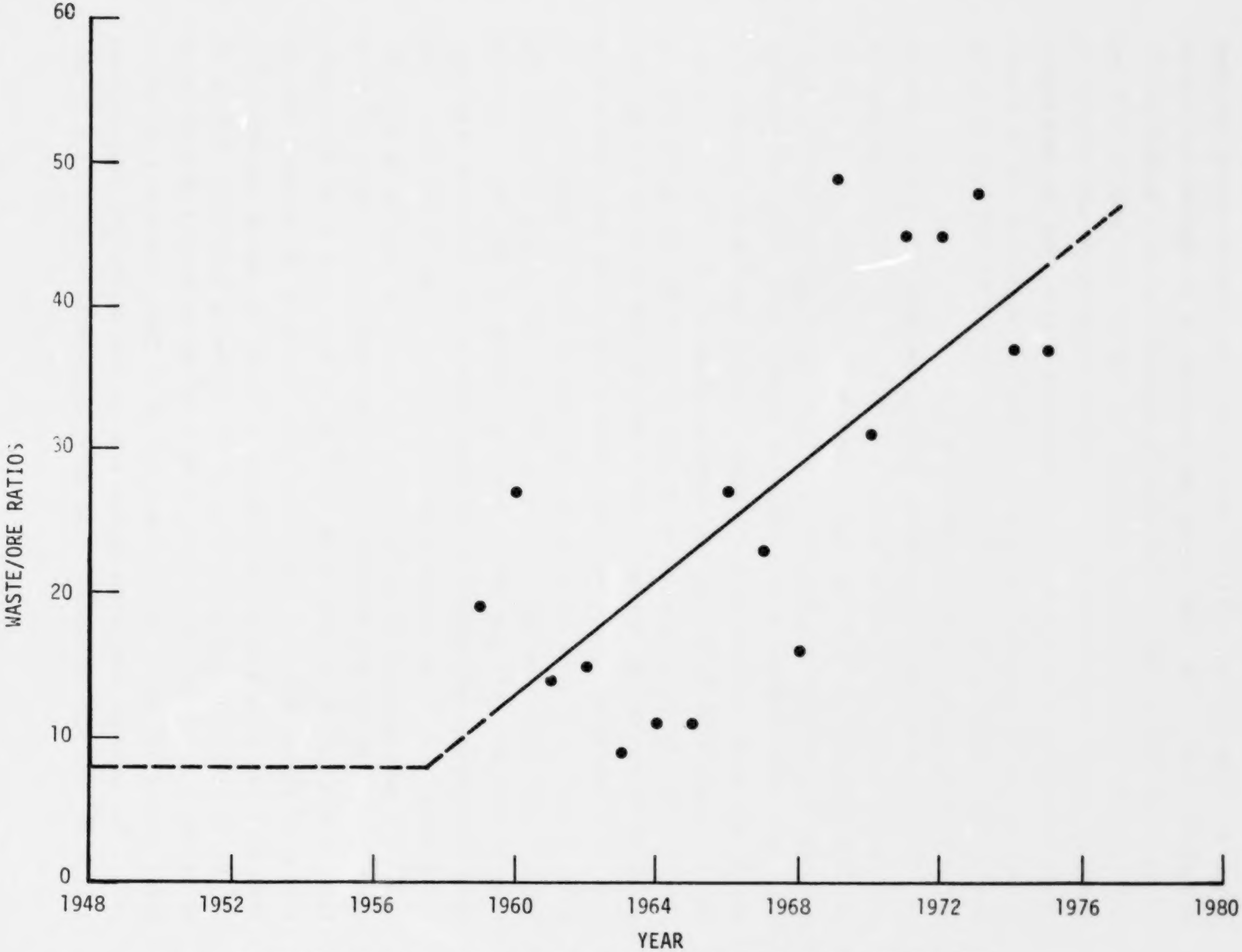

Figure 3.23 Annual waste to ore ratios for surface mining of uranium (1948 to 1979).

Since early surface mines recovered ore bodies very close to the surface, the ore-to-waste ratio would be expected to be relatively small. A range of waste to ore ratios of 8:1 to 35:1 for surface mining has been estimated (C174). The lower ratio was selected to be typical for surface mining from 1948 to 1957 and was used to estimate the waste production for that period. The increase in waste-to-ore ratios from 1959 to 1976 was probably due to several reasons. The gradual depletion of near surface ore deposits required mining deposits at increasing depths, and the development of surface mining equipment now permits economical recovery of ore at greater depths below grade. The waste-to-ore ratios for 1976 to 1977 were projected with the previous regression analysis line fit.

The estimated annual cumulative waste production from uranium surface mining for 1950 to 1978 (Table 3.69) is 1.73×10^9 MT. A crude estimate of the waste production for the model inactive surface mine can be made by dividing the total waste produced to 1978 by the number of inactive mines. But, this overestimates waste production because some of the contemporary wastes are being produced by active mines, and the waste production per mine has increased with increasing contemporary waste-to-ore ratios. To adjust the contemporary waste production for the active mines and the increasing waste-to-ore ratios, we assumed a cutoff date of 1970, based on the description of a contemporary active surface mine (Ni79). The model mine age is about 1 year as of June 1978, and has an expected life of approximately 17 years. Those mines that were active in 1970 are all assumed to have become inactive between 1970 and 1978. Their percentage of the annual waste of about 12.5 percent was assumed to decrease linearly with time from 1970-1978. For example, all of the wastes produced by surface mines in 1970 (i.e., 7.69×10^7 MT) were produced by surface mines that would be inactive by 1978. The waste production for the following years (1971-1977) was: 1.05×10^8 MT in 1971; 1.16×10^8 MT in 1972; 1.14×10^8 MT in 1973; 6.49×10^7 MT in 1974; 5.24×10^7 MT in 1975; 4.77×10^7 MT in 1976; 2.97×10^7 MT in 1977. The ore production was calculated in the same manner as for the wastes and was 3.27×10^7 MT in 1970. The ore production for the following years was: 2.32×10^6 MT in 1971; 2.58×10^6 MT in 1972; 2.38×10^6 MT in 1973; 1.76×10^6 MT in 1974; 1.43×10^6 MT in 1975; 1.06×10^6 MT in 1976, and 6.32×10^5 MT in 1977. The adjusted cumulative wastes from surface mining from 1950-1978 was 1.11×10^9 MT, and the adjusted cumulative ore production was 4.49×10^7 MT.

Table 3.69 Cumulative uranium mine waste and ore production

Year	Waste (10^3MT)		Ore (10^3MT)	
	Surface	Underground	Surface	Underground
1977	1733000	29250	59220	73100
1976	1496000	24950	54160	68840
1975	1305000	21380	49920	65210
1974	1165000	19180	46110	62760
1973	1036000	17760	42600	60500
1972	853200	16820	38800	58960
1971	698100	16240	35350	56510
1970	577800	15370	32700	53600
1969	501000	14410	30200	50330
1968	420000	13220	28550	47160
1967	388600	12060	26560	43810
1966	356200	11040	25170	40910
1965	331800	10180	24260	38090
1964	314000	9369	22640	35000
1963	287300	8425	20290	31750
1962	254200	7479	16720	28210
1961	209600	6391	13810	23310
1960	167100	5273	10770	18320
1959	93510	3406	8075	13150
1958	46720	2466	5580	9433
1957	27470	1776	3442	6839
1956	15770	1266	1979	4943
1955	6720	852	848	3356

Table 3.69 (Continued)

Year	Waste (10^3MT)		Ore (10^3MT)	
	Surface	Underground	Surface	Underground
1954	4071	580	509	2313
1953	370	171	46.3	702
1952	842	257	105	1043
1951	370	171	46.3	702
1950	167	98.9	20.9	413
1949		48.6		206
1948		11.4		49.8
1947		3.18		15.3
1946		3.18		15.3
1945		3.18		15.3
1944		3.18		15.3
1943		3.18		15.3
1942		3.18		15.3
1941		3.18		15.3
1940		3.00		14.5
1939		2.85		13.8
1938		1.67		8.12
1937		0.844		4.24
1936		0.533		2.68
1935		0.272		1.37
1934		0.0656		0.333
1933		0.0195		0.102
1932		0.0105		0.0553

Using these adjusted waste and ore values, the model inactive uranium surface mine produced 8.88×10^5 MT of waste and 3.59×10^4 MT of ore.

The volume of the remaining pit of the model surface mine would be equal to the total of the volume of wastes and ore that were removed from the mine. Assuming a density of 2.00 MT/m^3, the volume of wastes and ore removed from the mine pit would be 4.44×10^5 and 1.80×10^4 m^3, respectively. The pit was assumed to have the shape of an inverted truncated cone with a wall angle of 45° (Fig. 3.24). The ore body was assumed to be a solid right cylinder with a radius of 43.7 m and height of 3.0 m. The pit depth (ground surface to bottom of ore bed) was 36.7 m, and the ground surface area of the pit opening was calculated to be 2.03×10^4 m^2.

3.7.1.1 Waste Rock Piles

Overburden and sub-ore wastes from surface mines have been handled in several ways in the past. In one case, the sub-ore (generally the last material removed from the pit) was piled on top of the overburden. In another case the sub-ore was piled separately and blended with higher grade ore for shipment to the ore buying stations or mills. If the quantity of sub-ore was in excess of that required for blending, it was also dumped on top of the overburden (personal communication with G. Ritter, Bendix Field Engineering Corp., Grand Junction, CO, 1979). The earlier surface mining practices, therefore, generally produced waste piles with their cores containing overburden and their outer surface containing a mixture of overburden and sub-ore.

The actual method of removing and stacking overburden and sub-ore varies from mine to mine. In many cases the wastes were dumped in depressions or washes or stacked in more than one pile. For calculation purposes, we assume that wastes are stacked on a single pile in the shape of a solid truncated cone 10 m high with a 45 degree slope. It is further assumed that the sub-ore removed from the pit is placed evenly on top of the stacked overburden. The area and depth of the sub-ore placed on the waste pile is estimated by determining the areas of the base and top of the pile by iteration, computing the exposed surface area of the pile, computing the volume of the sub-ore, and calculating the depth of the sub-ore.

The areas of the base and top of the waste pile (truncated cone) were determined from the following equation:

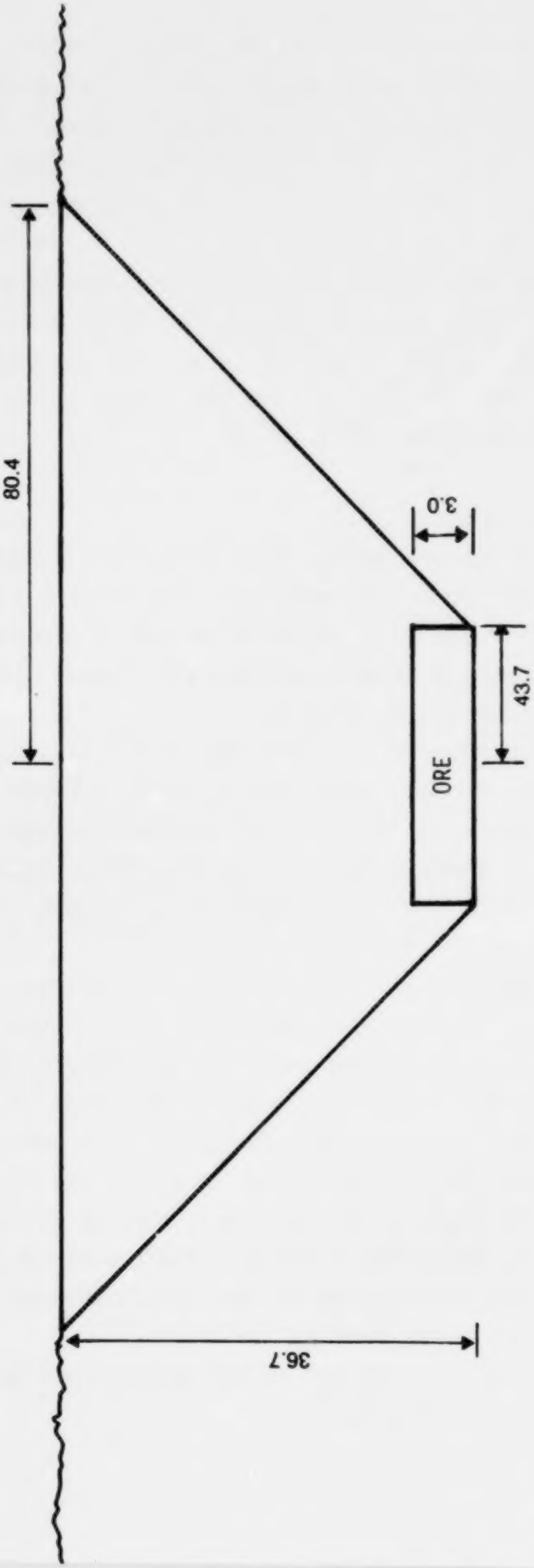

Figure 3.24 Cross section of model inactive surface mine (meters).

$$V = \frac{h}{3}\left(A_B + A_T + \sqrt{A_B A_T}\right) \qquad (3.25)$$

where V = volume of wastes (overburden and sub-ore) (m^3)
A_B = area of the base (m^2)
A_T = area of the top (m^2)
h = perpendicular distance between the base and top (10 m)

Different values of A_B were substituted into the equation until the value of V was equal to the combined volumes of the overburden and sub-ore (i.e; $5.55 \times 10^5\ m^3$) using a bulking factor of 25%. The area of the cone top was computed (assuming a 45 degree slope) from the diameter of the top (D_T), which is equal to the diameter of the base (D_B), minus 20 meters or $D_T = D_B - 20$. The calculated diameters, D_T and D_B, are 256 m and 276 m, respectively.

The exposed surface area of the waste pile was calculated using the following equation:

$$S = S_L + S_T \qquad (3.26)$$

where S_L = lateral surface area (m^2)
$S_L = \frac{L}{2}(C_B + C_T)$
and S_T = area of the top (m^2)
$= \pi r_T^2$

$$S = \frac{L}{2}(C_B + C_T) + \pi r_T^2$$

where C_B = circumference of the base (m)
C_T = circumference of the top (m)
L = slant height (m)
r_T = radius of the top (m)

$$S = \frac{14.1}{2}(\pi D_T + \pi D_B) + \pi r^2$$

where D_T = diameter of the top (m)
D_B = diameter of the base (m)

$$S = \frac{14.1}{2}(3.14)(256 + 276) + 3.14\,(16384)$$

$S = 6.33 \times 10^4\ m^2$ (exposed surface area of waste pile)

The volume of sub-ore removed from the pit is assumed to be equal to the volume of ore removed from the pit. The thickness (T) of the sub-ore plate on the overburden is--

$$T = \frac{Vo}{S} = \frac{2.25 \times 10^4 m^3}{6.33 \times 10^4 m^2} = 0.36m. \qquad (3.27)$$

In summary, the waste pile produced at an inactive uranium surface mine is to be in the shape of a truncated cone having a surface area of 6.33×10^4 m^2. The pile is assumed to have an inner-core of overburden plated with 0.36 m of sub-ore on its exposed surface. In practice, the plate would be a mixture of overburden and sub-ore with the sub-ore concentrations increasing towards the pile surface.

Table 3.70 lists average annual emissions of contaminants due to wind erosion of the overburden pile. To compute these values, an emission factor of 0.850 MT/hectare-yr, computed in Appendix I, was multiplied by the pile surface area, 6.33 hectares, and the stable element concentrations listed in Table 3.19. Uranium and thorium concentrations were assumed to be 110 pCi/g and 2 pCi/g, respectively.

3.7.1.2 Radon-222 from the Mine Area

After the termination of active mining, Rn-222 will continue to exhale from the wall and floor of the pit. Since all of the ore has been removed, the Rn-222 will originate from the overburden and sub-ore surfaces. The surface area of the sub-ore region of the pit is estimated from the volume of ore and sub-ore (3.6×10^4 m^3) and the shape and size of the pit using the following equations:

$$V = 1/3\ h\ (A_T + A_B + \sqrt{A_T A_B})\ \text{where: } A_T = \pi r_T^2 \qquad (3.28)$$

$$S = 1/2\ L\ (C_B + C_b) + A_B \qquad A_B = \pi r_B^2 \qquad (3.29)$$

The terms in the equations are defined in the previous Section. By substituting the terms $r_B + h$ for r_T in Equation 3.28, h can be solved by iteration.

$$V = 3.6 \times 10^4\ m^3 = 1/3\ h\ [\pi\ (r_B + h)^2 + \pi r_B^2 + \sqrt{\pi\ (r_B + h)^2\ (\pi r_B^2)}\ ]\ \text{when } h = 5.3\ m$$

The exposed surface area of the pit that contains the sub-ore is

$$S_s = 1/2\ (7.50)\ \pi(87.4 + 98.0) + 6000 = 8.18 \times 10^3\ m^2,$$

and the surface area of the overburden section of the pit is

$$S_o = 1/2\ (44.4)\ (\pi)\ (98.0 + 161.0) = 1.81 \times 10^4\ m^2.$$

Table 3.71 shows the results of radon flux measurements made at 20 of the tailings piles at inactive uranium mill sites. Also shown is the estimated average Ra-226 content of the tailings and the average Ra-226 content

Table 3.70 Average annual emissions of radionuclides (μCi) and stable elements (kg) in wind suspended dust at the model inactive surface mine

Contaminant	Overburden Pile[a]	Contaminant	Overburden Pile[a]
Arsenic	0.46	Molybdenum	0.62
Barium	4.9	Nickel	0.11
Cadmium	ND[b]	Lead	0.42
Cobalt	0.09	Ruthenium	ND
Copper	0.33	Selenium	0.59
Chromium	0.11	Strontium	0.70
Iron	84	Vanadium	7.6
Mercury	ND	Zinc	0.16
Potassium	135	Uranium-238 and each daughter	1480
Magnesium	19	Thorium-232 and each daughter	11
Manganese	5.2		

[a] Mass Emissions = 5.38×10^6 g/yr.
[b] ND - Not detected.

Table 3.71 Average radon flux of inactive uranium mill tailings piles

Location	Average Radon Flux(a) (pCi/m^2-sec)	Estimated Ra-226(b) Tailings Content (pCi/g)	Average Ra-226 Background Soils(a) (pCi/g)	Reference(a)
ARIZONA				
Monument Valley	20	50	0.95	FBD-GJT-4 (1977)
Tuba City	193	924	0.95	FBD-GJT-5 (1977)
COLORADO				
Durango	197	840	1.48	FBD-GJT-9 (1977)
Grand Junction	359	784	1.52	FBD-GJT-9 (1977)
Gunnison	470	420	1.48	FBD-GJT-12 (1977)
Maybell	86	252	1.52	FBD-GJT-11 (1977)
Naturita	1446	756	1.48	FBD-GJT-8 (1977)
New Rifle	458	504	1.52	FBD-GJT-10 (1977)
Old Rifle	553	980	1.52	FBD-GJT-10 (1977)
Slick Rock	70	171	1.48	FBD-GJT-7 (1977)
IDAHO				
Lowman	125	---	1.12	FBD-GJT-17 (1977)
NEW MEXICO				
Ambrosia Lake	173	760	1.02	FBD-GJT-13 (1977)
Shiprock	340	700	1.7	Bernhardt et al. (1975)
OREGON				
Lakeview	660	420	0.81	FBD-GJT-18 (1977)
SOUTH DAKOTA				
Edgemont	143	---	1.33	FBD-211 (1978)
TEXAS				
Falls City	65	448	0.93	FBD-GJT-16 (1977)
Ray Point	430	518	0.93	FBD-GJT-20 (1977)
UTAH				
Green River	77	140	1.43	FBD-GJT-14 (1977)
Mexican Hat	290	784	0.83	FBD-GJT-3 (1977)
Salt Lake City	1200	896	1.4	Bernhardt et al. (1975)
WYOMING				
Spook Site	1770	356	0.99	FBD-GJT-15 (1977)
Average All Sites	466	563	1.26	

(a) FBD77.
(b) Sw76.

measured in representative background soils for each site. The average radon exhalation rate per average Ra-226 content of tailings material from these data is 0.83 pCi of Rn/m^2-sec per pCi of Ra/g.

Data analysis by Schiager (Sc74) indicates a radon exhalation rate of 1.6 pCi of Rn/m^2-sec per pCi of Ra/g. This value has often been used in the environmental impact statements to assess the radon flux from tailings materials.

Table 3.72 summarizes data obtained during radiological surveys of inactive uranium mine sites in New Mexico and Wyoming during the spring of 1979 (Wo79). Radon exhalation rates were measured with charcoal cannisters and the radium-226 concentrations were determined for composite surface samples taken from overburden, sub-ore, and waste rock piles. The average radon-222 exhalation rate per average radium-226 content of the overburden, sub-ore, and waste rock piles was 0.27, 0.11, and 0.12 pCi of Rn/m^2-sec per pCi of Ra-226/g, respectively.

Measurements of the background flux and Ra-226 content of typical background soils were reported for the Edgemont, South Dakota site (FBD78). These data indicate a value of 1.05 pCi of Rn/m^2-sec per pCi of Ra/g. Table 3.73 summarizes background radon flux estimates for several regions of the United States. Considering the average U.S. background flux to be 0.82 pCi of Rn/m^2-sec (Tr79) and the average U.S. background soil Ra-226 content to be 1.26 pCi of Ra/g (Oa72), the average U.S. background radon exhalation rate is estimated to be 0.65 pCi of Rn/m^2-sec per pCi of Ra/g. The average background radon exhalation rate for New Mexico and Wyoming (Table 3.72) was 0.33 pCi of Rn/m^2-sec per pCi of Ra/g. Therefore, the grand average U.S. background radon exhalation rate has been estimated to be 0.68 pCi of Rn/m^2-sec per pCi of Ra/g, and the grand average U.S. background soil Ra-226 content has been estimated to be 1.6 pCi/g.

We estimated the total radon released from the model abandoned surface mine area from the following parameters:

1. Radon exhalation from the sub-ore surface area of the pit--
    - the exposed sub-ore surface area (S_s) = 8.18 x 10^3 m^2;
    - the average radium-226 content of the sub-ore = 110 pCi/g; and
    - the radon flux rate for sub-ore = 12 pCi of Rn/m^2-sec.

Table 3.72 Average radon flux measured at inactive uranium mine sites

Location	Area	Average Radon Flux (pCi/m^2-sec)	Number of Flux Measurements	Average Radium-226 Content of Surface Sample (pCi/g)
Underground Mines				
San Mateo Mine, New Mexico	Waste pile	18	11	117
	Heap leach pond	38	3	81
	Background	0.29	1	0.77
Barbara J # 1 Mine, New Mexico	Waste pile	7.9	6	110
	Background	0.41	1	3
Surface Mines				
Poison Canyon 1, New Mexico	Sub-ore	7.0	1	43
	Overburden piles	6.7	5	62
	Background	0.33	1	2.1
Poison Canyon 2, New Mexico	Sub-ore	5.3	3	---
	Overburden pile	9.8	6	---
Poison Canyon 3, New Mexico	Sub-ore	11	2	---
Morton Ranch (Pit 1601), Wyoming	Sub-ore	24	12	170
	Overburden	9.7	4	23
	Background	2.3	2	3
Grand Averages	Sub-ore	12		110
	Overburden	8.7		32
	Waste Rock	13		110
	Background	0.83		2.2

Source: Wo79.

Table 3.73 Background radon flux estimates

Location	Radon Flux pCi/m^2-sec
Background Soils of the U.S.	
Champaign County, Illinois	1.4
Argonne, Illinois	0.56
Lincoln, Massachusetts	1.3
Socorro, New Mexico	0.90
Socorro, New Mexico	1.0
Socorro, New Mexico	0.64
Yucca Flat, Nevada	0.47
Texas	0.27

Average U.S. Background Radon Flux = 0.82 pCi/m^2-sec.

Source: Tr79.

Therefore, the radon released from the sub-ore surface area of the pit is $8.18 \times 10^3\ m^2 \times 12$ pCi of Rn/m^2-sec x 86400 sec/day = 8.48 mCi of Rn/day.

2. Radon exhalation from the overburden surface area of the pit--
   - the exposed overburden surface area (S_o) = $1.81 \times 10^4\ m^2$;
   - the average radium-226 content of the overburden = 32 pCi/g; and
   - the radon flux rate for overburden is 8.7 pCi of Rn/m^2-sec.

Therefore, the radon released from the overburden surface area of the pit is $1.81 \times 10^4\ m^2 \times 8.7$ pCi of Rn/m^2-sec x 86400 sec/day = 13.6 mCi of Rn/day.

3. Radon exhalation from the overburden pile remaining at the pit--
   - the exposed surface area of the waste pile (S_w) = $6.33 \times 10^4\ m^2$;
   - the Ra-226 content of the surface of the overburden pile is the same as the sub-ore content = 32 pCi/g; and
   - the radon flux rate for the overburden pile is 8.7 pCi of Rn/m^2-sec.

Therefore, the radon exhalation rate from the overburden pile is $6.33 \times 10^4\ m^2 \times 8.7$ pCi of Rn/m^2-sec x 86400 sec/day = 47.6 mCi of Rn/day.

The total radon release rate at the abandoned surface mine site is the sum of the above three source terms, 69.7 mCi/day. The estimated radon release rate for background soils for an undisturbed area equivalent to the surface mine area uses the following parameters:

- the ground surface area equivalent to the area of the pit opening ($2.03 \times 10^4\ m^2$) and the overburden pad area ($5.98 \times 10^4\ m^2$) = $8.01 \times 10^4\ m^2$, and
- the radon flux rate for background soils in uranium mining areas = 0.83 pCi of Rn/m^2-sec (Table 3.72).

Therefore, the radon exhalation rate from an undisturbed area equivalent to the model surface mine is

$8.01 \times 10^4\ m^2 \times 0.83$ pCi of Rn/m^2-sec x 86400 sec/day = 5.7 mCi of Rn/day.

Table 3.74 summarizes the annual radon-222 release from the model inactive uranium surface mine and all inactive uranium surface mines.

3.7.1.3 Land Surface Gamma Radiation

The surface mine uranium overlying strata must be removed in order to gain access to the uranium-bearing host materials and the ore body. The ore body consists of ore and sub-ore, and the sub-ore is simply that fraction of the ore body that contains ore uneconomical to recover. The end result of the mining is that the residues (sub-ore) enhance natural radioactive materials. That is, they are exposed or brought to the earth's surface. The enhancement will cause, in most cases, increased aboveground radiation

Table 3.74 Summary of estimated radon-222 releases from inactive surface mines

Source	Estimation Method	Annual Release, Ci
Mine Pit		
Sub-ore area	Model mine and limited field measurements	3.1
Overburden area	Model mine and limited field measurements	5.0
Total		8.1
Overburden Pile	Model mine and limited field measurements	17.4
Background	Rn-222 flux measurements and projected surface areas of model mine pit and overburden pile	2.1
Model Mine	Net Rn-222 release	23.4
All Inactive Mines	Annual net Rn-222 release from model times 1250 mines	29,000

exposure rates around the mining area. Ore and sub-ore lost through handling are subject to wind and water erosion. This effectively increases the mine site area in a radiological sense. The gamma radiation exposure levels on and around a mine site can be high enough to restrict use of the area after mining.

Gamma radiation surveys were conducted at some inactive uranium surface mining areas. Table 3.75 lists the ranges of exposure rates found. Appendix G contains more specific information concerning the surveys. The residual exposure rate levels would probably preclude unrestricted use of the pits, waste piles, and overburden.

Figure 3.25 depicts gamma radiation measurements made on radials extending outward from an inactive surface mine pit. The measurements were made with a pressurized ion chamber (PIC) at approximately 61 m intervals on each radial. As expected, the exposure rate decreases with distance away from the pit, indicating surface contamination from wind and water erosion of the spoils and ore piles. Some of the contamination may also have originated from ore and sub-ore dust losses during mining.

Since the pit resides over a former ore body and connecting or adjacent ore bodies may be located near the mine, some caution is necessary when interpreting the gamma exposure rates as indicative of surface contamination. Development drilling, indicating the presence of ore bodies, is prevalent throughout the north, west, and south areas around the pit. The northeast, east, and southeast areas around the pit have exploratory drill holes only. They indicate the probable absence of ore bodies. Although the north, northwest, west, and southwest radials cross below grade ore bodies, it is not reflected by the gamma measurements. Unless the ore body is very close to the surface, its gamma radiation will not be measured (i.e., the 1/10 value layer for earth shielding is about 0.3 m). The south radial, however, did cross an ore outcropping.

If the exposure rate measurements made at the end points of the radials (south radial excepted) are assumed to be near background, their mean value is 14.4 μR/hr with a 2 sigma error of 1.6 μR/hr.

Assuming all measurements in excess of 14.4 + 1.6 μR/hr or 16.0 μR/hr are a result of eroded ore and sub-ore from the mining activities, an iso-exposure rate line enclosing the eroded materials can be constructed around the mine site. The line is constructed on Fig. 3.25 and is qualitatively

Table 3.75 Summary of land surface gamma radiation surveys in New Mexico, Texas and Wyoming

Location	Area	Gamma Radiation Exposure Rate (μ R/hr)
Poison Canyon, New Mexico	Pits	40 to 190
	Waste piles	65 to 250
	Overburden	25 to 65
Texas	Pits	5 to 400
Morton Ranch, Wyoming (1601 Pit)	Pit	16 to 63
	Ore piles	200
	Overburden	59 to 138

Source: Wo79 for New Mexico and Wyoming and Co77 for Texas.

adjusted on the south radial to compensate for the ore outcropping. The line bulges into the southeast quadrant indicating erosion by the predominant northwest winds and contamination of about 0.3 km^2.

In summary, it appears that the residual gamma radiation exposure levels at surface mining pits and overburden piles would preclude these areas from unrestricted use. It also appears that wind and water erosions of the spoils, ore, and sub-ore are occurring and causing land contamination far removed from the mining area. Several surface mines were gamma surveyed in New Mexico. The mines could not be individually gamma radiation surveyed because of their close proximity, cross contamination from eroded ore and sub-ore, and possible ore outcrops.

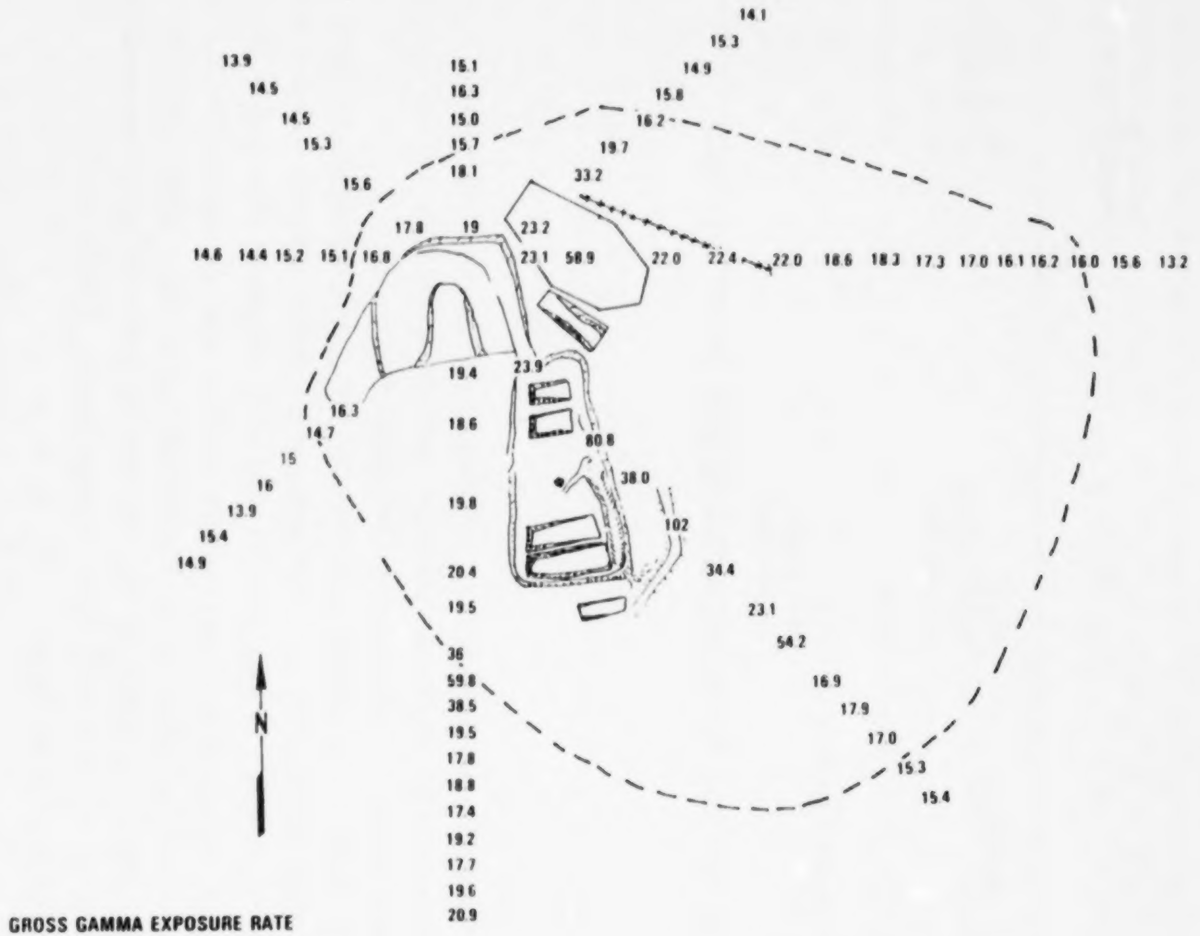

Figure 3.25 Results of gamma exposure rate survey at the 1601 pit and environs, Morton Ranch uranium mine, Converse County, Wyoming (μR/hr)

3.7.2 Inactive Underground Mines

The model inactive underground mine is basically defined by dividing the total reported volumes of ore and waste removed by inactive underground mining by the number of inactive underground mines. The number of inactive underground mines has been obtained from the U.S. Department of Energy mine listing in Table 3.67. Table 3.67 lists the mines by state and type of mine. Forty-four percent of the inactive underground mines are located in Colorado, 34 percent in Utah, 9.3 percent in Arizona, and 7.0 percent in New Mexico.

For modeling purposes, we assume that there are presently 2030 inactive underground uranium mines. Table 3.69 lists the estimated underground mine waste and ore production for 1932 to 1977. Uranium mine waste and ore production statistics, on an annual basis, were available for underground producers from 1959 to 1977 (DOI59-76). Annual uranium ore production statistics for underground mining are available from 1948 to 1959 (DOE79) and from 1932 to 1942 (DOI32-42). We estimated the mine waste production for the period of 1932 to 1960 from underground mining waste-to-ore ratios and established waste-to-ore ratios using the published ore and wastes production statistics from 1959 to 1976 (DOI59-76). These ratios were fitted with a line by regression analysis in order to estimate the waste-to-ore ratios from 1932 to 1959 (Fig. 3.26). Two lines were fitted to the known waste-to-ore ratios because of the abrupt change in the ratios from 1972 to 1976. We assumed that the steeper slope was caused by increased waste production from the larger and deeper underground mines operated during this time. The estimated annual waste-to-ore ratios were multiplied by the published annual ore production values to estimate the annual waste production from 1932 to 1959. We assumed that no ore was produced from 1942 to 1948 because most of the uranium was obtained by reprocessing vanadium and radium tailings during that period (Private communication with G. C. Ritter, 1979, Bendix Field Engineering Corporation, Grand Junction, Colorado). Table 3.69 lists the cumulative annual waste production from underground mining from 1932 through 1977. The total waste produced for this period was 2.92×10^7 MT, and the total ore produced was 7.31×10^7 MT.

A simplistic way to identify a model inactive underground mine would be to divide the cumulative tonnage of ore and wastes by the number of inactive mines. We estimated the number of inactive mines from the U.S. Department of Energy mine listing (Section 2.0 and Table 3.67). The model inactive under-

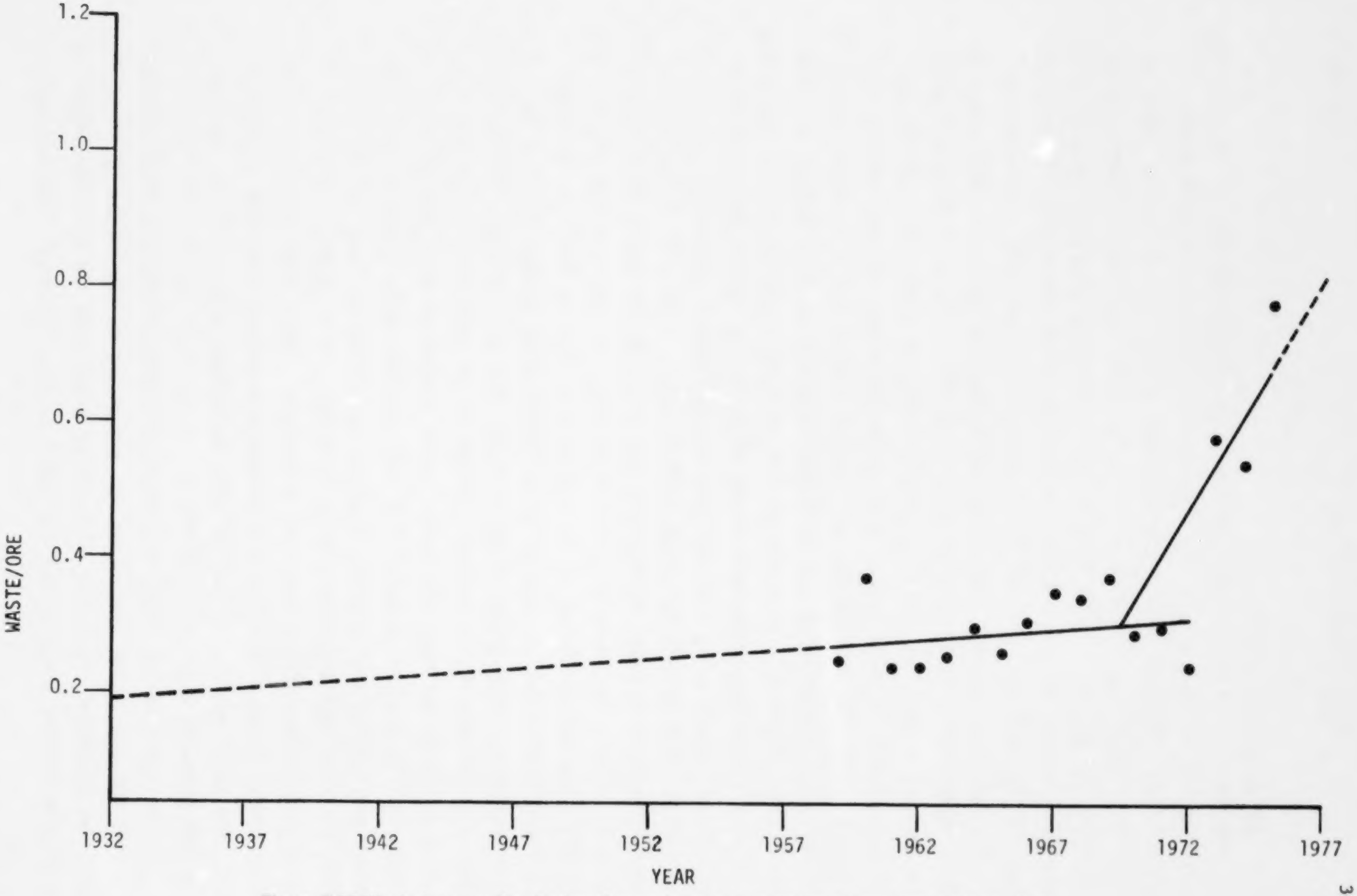

Figure 3.26 Waste to ore ratios for inactive underground uranium mines from 1932 to 1977.

ground mine produced 3.60×10^4 MT of ore and 1.44×10^4 MT of waste. Unfortunately, some of the contemporary waste and ore production has been produced by both active and inactive mines. In order to adjust the contemporary ore and waste production for that portion of the ore and wastes generated by active mining, we assumed a model active mine having a mining life of 15 years (St79). The mid-life of the mine was assumed to have occurred in 1978, with production beginning in 1971.

We also assumed that some of the mines became inactive during the 1971-1978 period and that their numbers decreased linearly. For example, 2.44×10^6 MT of ore was produced in 1972 and 85.7 percent of that ore produced was from mines that were inactive by 1978. Therefore, adjusted ore production was 2.09×10^6 MT for 1971. The ore production for 1973 was 1.15 $\times 10^6$ MT and 1.27×10^6 MT in 1974; 1.06×10^6 MT in 1975; 1.02×10^6 MT in 1976; and 6.16×10^5 MT in 1977. The adjusted waste production was: 5.08×10^5 MT in 1972; 6.67×10^5 MT in 1973; 8.13×10^5 MT in 1974; 9.43×10^5 MT in 1975; 7.43×10^5 MT in 1976; and 4.99×10^5 MT in 1977.

Through 1978, the cumulative adjusted ore production from inactive underground mines was 6.37×10^7 MT, and the cumulative adjusted waste production was 2.04×10^7 MT. The model inactive underground mine was assumed to have produced 3.14×10^4 MT of ore and 1.00×10^4 MT of waste. Assuming a density of 2.0 MT per m^3, the volume of ore and waste removed were 1.6×10^4 and 5.0×10^3 m^3, respectively.

Fifty percent of the waste volume mined we assumed to be sub-ore. The volume of waste rock (i.e., containing no sub-ore) removed during the mining is 2.5×10^3 m^3. Assuming an entry dimension of 1.83 m x 2.13 m, about 615 m of shafts and haulways are in the model mine. The ore body we assumed to have an average thickness of 1.8 m with a length and width of 91.2 m each. The surface area of the passages would be 4.83×10^3 m^2. The surface area of the mined-out ore body would be 1.71×10^4 m^2.

3.7.2.1 Waste Rock Piles

Wastes produced from underground uranium mining were generally cast or dumped near the mine entries. Those wastes that were dumped on relatively flat terrain formed dome-shaped piles. Wastes cast from rim mines generally formed long, thin sheets down the canyon slopes. Since most of the inactive underground mines are in the Uravan Mineral Belt, the waste pile shape (dome) is assumed to be predominant (see Appendix G.1.2) and is used for the calculations of the waste pile dimensions.

The waste produced at a typical underground mine consists of waste rock and sub-ore. The waste rock is assumed to be on the bottom of the waste pile since it was generally removed first. Sub-ore, which was removed later, is assumed to cover or plate the waste pile. The waste piles are assumed to be dome shaped, covering a circular area of 0.40 hectares. The dome is assumed to be a spherical segment with a height (b) and base (c) of 71.8 m. The volume (V) of the spherical segment, 6.3×10^3 m^3 when corrected for bulking, is equal to the volume of wastes and is expressed as

$$V = \frac{1}{24} \pi b (3c^2 + 4b^2). \qquad (3.30)$$

The surface area of the spherical segment is given by the expression

$$S = \frac{1}{4} \pi (4b^2 + c^2) \text{ where } S = \text{Surface area } (m^2). \qquad (3.31)$$

The term b is solved by substitution and iteration in the former equation and is substituted in the latter equation to determine the surface area of the wastes:

$$V = 6.3 \times 10^3 \text{ m}^3 = \frac{1}{24} \pi \text{ b } (15465 + 4b^2) \text{ where: } b = 3.1 \text{ m}. \qquad (3.32)$$

The surface area of the waste pile is

$$S = \frac{1}{4} \pi (4b^2 + c^2) \qquad (3.33)$$

$$= \frac{1}{4} (3.14)(38 + 5155)$$

$$= 4.08 \times 10^3 \text{ m}^2.$$

The thickness (T) of sub-ore on the surface of the waste pile is

$$\frac{\text{volume of sub-ore}}{\text{area of waste pile}} = \frac{3.2 \times 10^3 \text{ m}^3}{4.08 \times 10^3 \text{m}^2} = 0.78 \text{ m}. \qquad (3.34)$$

In summary, the waste pile at an inactive underground uranium mine is assumed to have the shape of a spherical segment with a surface area 4.08×10^3 m^2. The pile is assumed to have an inner core of waste rock covered or plated with 0.78 m of sub-ore on its exposed surface. It is expected that the plate of sub-ore on the waste pile would be more pronounced than the sub-ore plates on overburden piles at surface mines because of diminished blending, mining practices, and the lower waste-to-ore ratio. The grand average of the radium-226 concentrations in the waste rock and overburden piles (Table 3.72) appear to confirm this expectation.

Table 3.76 lists average annual emissions of contaminants due to wind erosion of the waste rock pile. These values were estimated by multiplying an emission factor of 2.12 MT/hectare-yr, derived in Appendix I, by the waste pile surface area, 0.408 hectares, and the stable element concentrations given in Table 3.19. We assumed uranium and thorium concentrations to be 110 pCi/g and 2 pCi/g, respectively.

Table 3.76 Average annual emissions of radionuclides (μCi) and stable elements (kg) in wind suspended dust at the model inactive underground mine

Contaminant	Waste Rock Pile[(a)]	Contaminant	Waste Rock Pile[(a)]
Arsenic	0.07	Molybdenum	0.10
Barium	0.80	Nickel	0.02
Cadmium	ND[(b)]	Lead	0.07
Cobalt	0.01	Ruthenium	ND
Copper	0.05	Selenium	0.10
Chromium	0.02	Strontium	0.11
Iron	14	Vanadium	1.2
Mercury	ND	Zinc	0.03
Potassium	22	Uranium-238 and each daughter	238
Magnesium	3.0		
Manganese	0.83	Thorium-232 and each daughter	1.7

(a) Mass emissions = 8.65×10^5 g/yr.

(b) ND - Not detected.

3.7.2.2 Radon-222 from the Mine Area

We estimated the total radon released from the model inactive underground mine from the following parameters:

1. Radon exhalation from the waste rock pile--
   - the exposed surface area of the waste pile = 4.1×10^3 m^2;
   - the average Ra-226 content of the waste pile is 110 pCi/g; and
   - the radon flux rate for the waste pile is 13 pCi of Rn/m^2-sec.

Therefore, the radon released from the waste pile is

$$4.1 \times 10^3 \text{ m}^2 \times 13 \text{ pCi of Rn/m}^2\text{-sec} \times 86400 \text{ sec/day} = 4.6 \text{ mCi of Rn/day.}$$

2. Typical background release rate--
   - the ground surface area equivalent to the area covered by the waste pile = 4.1×10^3 m^2, and
   - the radon flux rate for background soils in uranium mining areas = 0.83 pCi of Rn/m^2-sec (Table 3.72).

Therefore, the radon exhalation rate from an undisturbed area equivalent to the waste pile of a model underground mine is

$$4.1 \times 10^3 \text{ m}^2 \times 0.83 \text{ pCi of Rn/m}^2\text{-sec} \times 86400 \text{ sec/day} = 0.29 \text{ mCi of Rn/day.}$$

The net radon release rate due to the waste pile at the inactive underground mine is 4.6 minus 0.29 or about 4.3 mCi of Rn/day above normal background.

Natural ventilation will occur in most mines and usually is considered by mine ventilation engineers when planning the forced ventilation systems. The natural force that can maintain a natural air flow due to temperature differences is thermal energy. The thermal energy added to a system is converted into a pressure difference. If the pressure difference is sufficient to overcome head losses, a flow of air will occur.

Natural ventilation depends upon the difference between the temperature inside and outside of a mine and the difference between the elevation of the mine workings and the surface. Air flow by natural ventilation is generally small (140 - 566 m^3/min) in shallow mines (Pe52). In deep mines, natural ventilation flows may range from 1,420 to 4,250 m^3/min (Pe52). The flow in either the shallow or deep mines depends upon the depth, size, and number of openings. The intensity of thermal energy-induced natural pressure usually ranges from a few hundredths to a few tenths cm of water in shallow (less than 460 m deep) mines (Pe52). The maximum pressure drop per 305 m of depth in deep mines is about 2.54 cm of water in winter and about 0.84 cm during the summer (Pe52).

In general, natural ventilation is subject to considerable fluctuation. It usually increases to a maximum in winter and a minimum in summer for deep mines. The typical inactive underground uranium mine would be shallow; therefore, the natural ventilation would be expected to reach its maximum in the winter and summer and its minimum in the spring and fall (air temperature in the mine closely approaches the outside temperature during the spring and fall).

A first approximation of the annual release of Rn-222 from an inactive underground mine simply would be that all Rn-222 released into the mine air will be exhausted by natural ventilation before a significant radioactive decay occurs. That is, the quantity of radon released into the mine is equal to the quantity of radon released from the mine. The quantity of Rn-222 released from the mined-out ore body is

$$Q\ \frac{(\text{pCi Rn-222})}{\text{sec}} = A \times \phi_{so}, \qquad (3.35)$$

where A is the surface area of the mined out ore body (m^2)

ϕ_{so} = exhalation rate of the Rn-222 from sub-ore per unit area per unit time, 12 $\frac{\text{pCi}}{m^2\text{-sec}}$ (Section 3.7.1.2)

$$Q = (1.71 \times 10^4\ m^2)\ \frac{(12\ \text{pCi})}{m^2\text{-sec}} = 2.1 \times 10^5\ \text{pCi/sec}.$$

It should be noted that ϕ_{so} is the average radon flux physically measured from sub-ore bodies in inactive surface mines (Section 3.7.1.2). Because of safety considerations, no measurements were made from sub-ore bodies in inactive underground mines during the April 1979 field surveys. The annual Rn-222 source term from the mined-out ore body in an inactive underground uranium mine, using the preceding assumptions, is

$$Q\ \frac{(\text{Ci})}{\text{yr}} = 2.1 \times 10^{5}\ \frac{\text{pCi}}{\text{sec}} \times 3.6 \times 10^{3}\ \frac{\text{sec}}{\text{hr}} \times 24\ \frac{\text{hr}}{\text{d}} \times 365\ \frac{\text{d}}{\text{yr}} \times \frac{1}{10^{12}\ \frac{\text{pCi}}{\text{Ci}}} = 6.6\ \frac{\text{Ci}}{\text{yr}}\ . \qquad (3.36)$$

The annual Rn-222 source term (Q) from the passageways, assuming an exhalation rate of 8.7 $\frac{\text{pCi}}{\text{m}^2\text{-sec}}$ for overburden, is (Section 3.7.1.2)

$$(4.8 \times 10^{3}\ \text{m}^{2})\ (2.7 \times 10^{-4}\ \frac{\text{Ci}}{\text{yr-m}^{2}}) = 1.3\ \frac{\text{Ci}}{\text{yr}}. \qquad (3.37)$$

The air flow rate from the mine, assuming 140 m^3/min for an average shallow mine, will exchange the mine air every three hours. The average annual radon-222 concentration will be

$$7.9\ \text{Ci/yr} \times \frac{1}{7.4 \times 10^{7}\ \text{m}^{3}/\text{yr}} \times \frac{1}{1000\ \ell/\text{m}^{3}} = 107\ \text{pCi}/\ell.$$

The radon daughter concentration will be about 87 percent of equilibrium with the radon, assuming a mean residence time of the radon in the mine to be 1.5 hours.

Several inactive mines in the Grants, New Mexico area were monitored for radon discharges by natural ventilation. One of the mines monitored was relatively small and had a vertical shaft access. Five cased 30 cm diameter vents were found and were assumed to be connected with the mine. The shaft was covered with steel plate, but access holes were cut in the plate and one corner had been pried up. Four vents were capped with buckets. Just one cover was gas tight. One vent was partially covered with a piece of wood. Only very small flow rates due to natural ventilation were measured at the shaft and vents. The maximum radon emission from the mine per day was estimated to be 2.8×10^3 μCi. This low radon discharge rate is probably due to partial blockage of the vents and water in the mine. The mine was partially flooded, and flowing water could be seen at the bottom of the shaft. The effect of the water would be to partially or completely close off the mine workings and substantially reduce natural ventilation. The water would also dissolve and substantially suppress the radon exhaling from the surface areas of the mine. Thus, we believe that the radon discharges from wet inactive mines via natural ventilation will be minimal.

Investigation at another inactive mine revealed that it was connected to three other inactive mines that were subsequently connected to two active mines. Ventilation fans at the connecting active mines were usually shut down after the end of the day shift and on weekends. Mine air was exhausted by natural ventilation through the shaft (highest opening) and vents of the mine investigated. A flow rate up to 88 m^3/min was observed coming from the shaft, and radon-222 concentrations reached 11,000 pCi/ℓ. The average flow rate observed over a weekend was 75 m^3/min, with an average radon-222 concentration of 9,800 pCi/ℓ. The average radon emission was 1.1 Ci/day.

Figure 3.27 is a plot of the changes in the Rn-222 concentration in the air from the shaft of the inactive mine investigated. The average of the measurements of the air flow rate from the shaft was about 76 m^3/min. The Rn-222 concentration increased almost linearly with time for about 20 hours after the fans were shut down at the end of the day shift on April 27, 1979. The Rn-222 concentrations also leveled off at about 10,000 pCi/ℓ. A dip, presumed to have been caused by high winds, occurred in the Rn-222 concentration curve from about 1000 to 1600 hours on April 28, 1979.

Since the curve is relatively flat at 10,000 pCi/ℓ, it is assumed that the rate of production of the Rn-222 is equal to the rate of removal of the Rn-222 from the six mines. The average residence time of the radon in the mine air is assumed to be approximately 10 hours, and the radon daughters would be in near-equilibrium (assumed to be ≅ 90 percent). Assuming that all six interconnecting mines contributed equally to the source term measured, the release rate of Rn-222 for a single mine will be

$$10{,}000 \text{ pCi/ℓ} \times 76{,}000 \text{ ℓ/min} \times 1440 \text{ min/day} \times 10^{-12} \text{Ci/pCi} \div 6$$
$$= 0.18 \text{ Ci/day.}$$

Based on the preceding estimation of Rn-222 and progeny released from a typical mine on the Colorado plateau and physical measurements at six connected mines, the annual radon release rate may range from 7.9 to 66 Ci/yr. These source term estimates, of course, are based on a single mine. Many mine workings are, in fact, interconnected. If these interconnected workings are assumed to constitute a single mine, then the upper limit of Rn-222 and

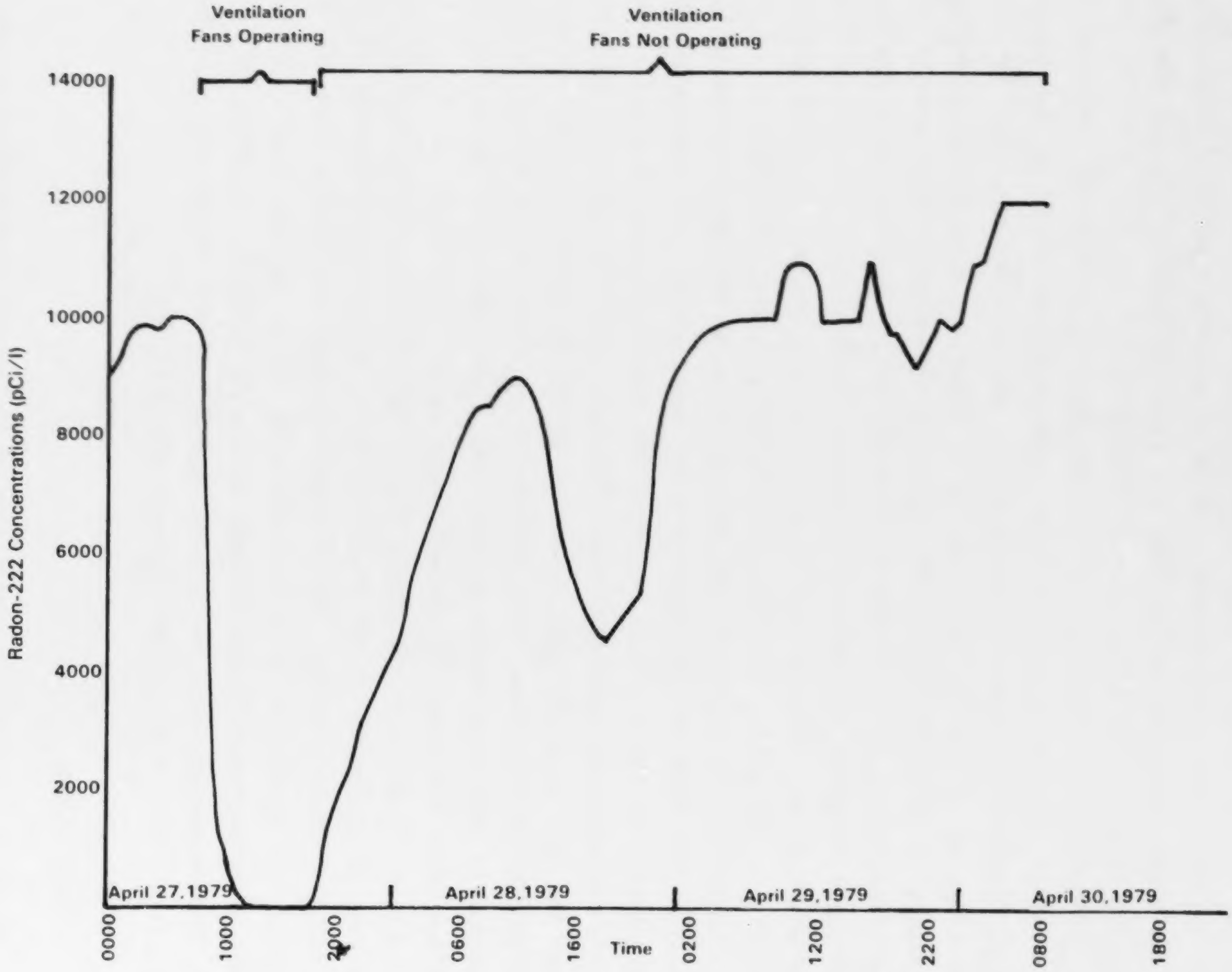

Figure 3.27 Radon-222 concentrations in mine air discharged by natural ventilation.

progeny discharge known at this time will be about 10,000 pCi/ℓ with an annual Rn-222 source term of about 400 Ci/yr. For example, 67 percent of all inactive underground uranium mines are in or near the Uravan mineral belt and are probably dry. Their aggregate Rn-222 discharge by natural ventilation is estimated to be

$$1360 \text{ mines} \times 66 \text{ Ci } \frac{\text{Rn-222}}{\text{yr-mine}} = 9.0 \times 10^4 \text{ Ci/yr.}$$

In summary, there is little information available on the discharge of Rn-222 and its progeny from the vents and entries of inactive uranium mines by natural ventilation. Some physical measurements indicate that the discharges may be substantial. It is known, through surveys conducted to support this study, that a large majority of the inactive uranium mines are not isolated from the atmosphere and are capable of discharging their Rn-222 and progeny into the local environment. It is also known that some self-sealing will probably occur at some of the mines, due to flooding, cave-ins, and subsidence. Table 3.77 summarizes estimates of the annual radon-222 releases from inactive underground uranium mines. This potential source of exposure could be practically eliminated by proper sealing of the inactive mines.

3.7.2.3 Land Surface Gamma Radiation

Gamma radiation surveys were conducted around underground mining areas in Colorado and New Mexico. Table 3.78 lists the ranges of gamma radiation exposure rates measured at some of the mines. The elevated gamma ray exposure rates on the waste piles are due primarily to plating those piles with sub-ore removed during the mining process.

Some radioactive materials originating from ore and sub-ore handling can be lost into the local environment around a mine site. Erosion of the mine wastes can also disperse contaminants into the local environment. Figure 3.28 illustrates gross gamma radiation exposure rate measurements around an inactive underground uranium mine in New Mexico. Background gamma-ray exposure rate measurements made around the mine area ranged from 12 to 15 μR/hr. According to the measurements made, exposure rate levels exceeded background from 50 to more than 100 meters from the waste piles. The area that has been contaminated far exceeds the area physically disturbed at the mine site. Gross gamma exposure rates measured on the waste piles averaged about 95

Table 3.77 Summary of radon-222 releases from inactive underground mines

Source	Estimation Methods	Annual Release, Ci
Model Mine		
Waste Rock Piles	Calculated volume & surface area; limited field measurements of radon flux	1.7
Underground workings	Radon release based on natural ventilation rate for shallow mines	
Sub-ore Surfaces	Calculated surface area; limited radon flux measurements of sub-ore	6.6
Passageways	Calculated passageway surface area; limited measurements of radon flux from overburden	1.3
Background	Field measurements of radon flux; and projected area of waste rock pile	0.11
Model Mine	Total radon source minus background	9.5
Actual Mine		
Underground workings (dry)	Field measurements	66
Underground workings (wet)	Field measurements	1.1
Waste rock piles	Calculated volume & surface area; Limited field measurements of radon flux	1.7

Table 3.78 Summary of land surface gamma radiation surveys in Colorado and New Mexico

Location	Area	Gamma Radiation Exposure Rate (μ R/hr)
Boulder, Colorado	Waste piles	40 to 100
Uravan, Colorado	Waste piles	50 to 220
San Mateo, New Mexico	Waste pile	35 to 275
	Ore	100 to 350
	Overburden	20 to 120
	Background	10 to 13
Mesa Top Mines, New Mexico	Waste piles	25 to 290
Barbara J #1 Mine, New Mexico	Waste piles	21 to 170
	Background	12 to 15

Source: Wo79.

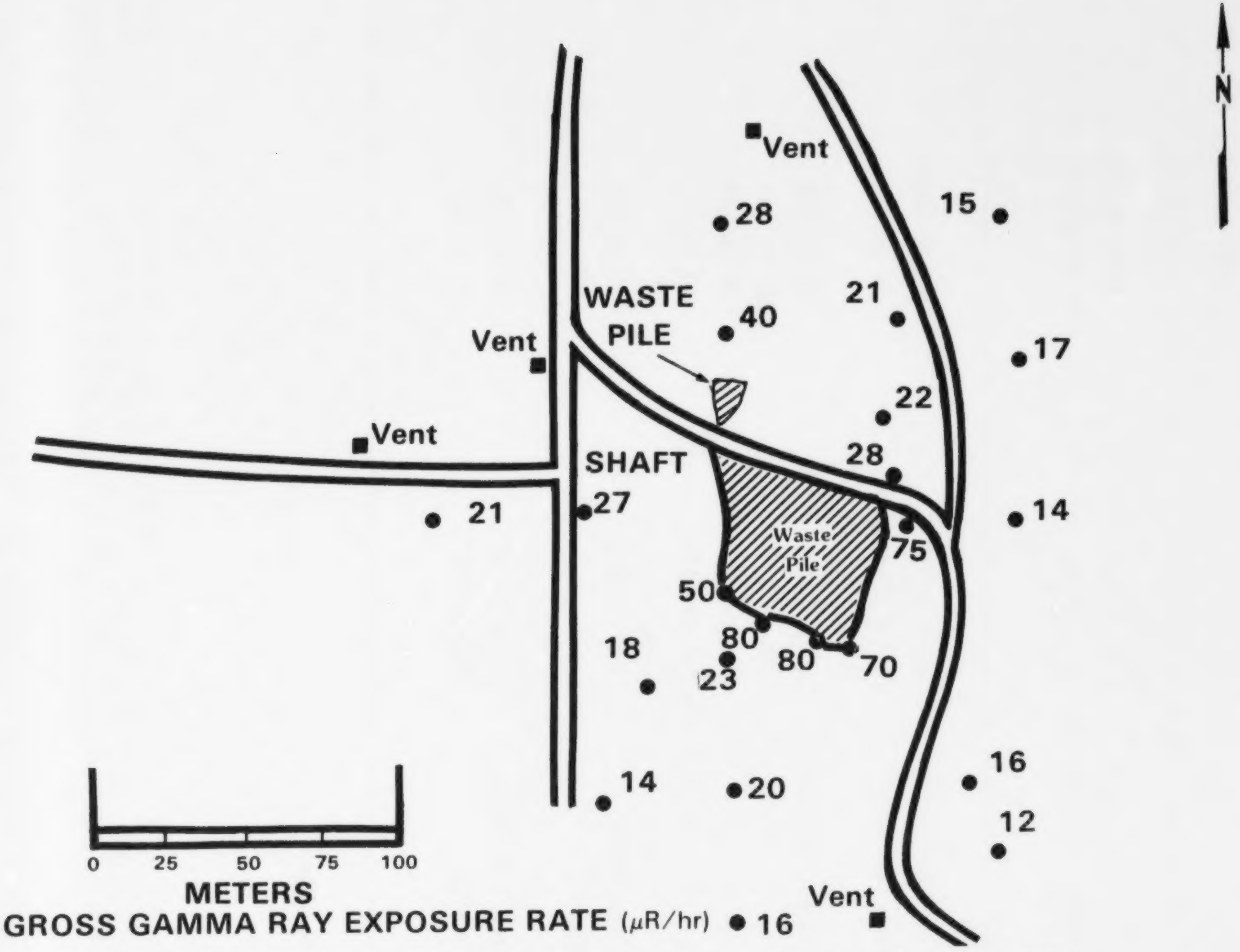

Figure 3.28 Gamma radiation survey around an inactive underground uranium mine in New Mexico.

μR/hr, which would make them unsuitable for unrestricted use.

In summary, wastes from underground uranium mining technologically enhance natural radioactivity and may be considered low-level radioactive wastes. Improperly controlled wastes will be dispersed into the surrounding environment by the mining activities and erosion.

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SECTION 4

DESCRIPTION OF MODEL MINES

4.0 Description of Model Mines

Section 1.3 describes uranium mines and their operations, and Section 3 describes the potential sources of contamination at the principal types of active and inactive mines. These discussions include an analysis of the potential sources of contamination, quantities of contaminants associated with the different sources, variations in the sources, and estimates of the values needed to define the impact that these sources may impose upon the environment and nearby populations. We attempted to define these terms and mining parameters in a way that would reflect a general view of the uranium mining industry and permit a generic assessment. The parametric values that we have chosen for this assessment are listed below. The sections of this report from which they were derived are given in parentheses.

4.1 Surface Mine

The model open pit (surface) mine will be located in Wyoming. It is the mine defined in Section 3.3 as the "average large mine." However, to define the total impact of all 63 open pit mines operating in the United States in 1978 we used the parameters developed in Section 3.3 for the "average mine."

Production Parameters (1.3.1, 3.3.1)

Parameter	Average Large Mine	Average Mine
Ore, MT/yr	5.1×10^5	1.2×10^5
Sub-ore, MT/yr	5.1×10^5	1.2×10^5
Overburden, MT/yr	4.0×10^7	6.0×10^6

Mining Parameters (3.3.1)

Parameter	Average Large Mine	Average Mine
Mining days per year	330	330
Mine life, yr	17	17
Ore stockpile residence time, days	41	41
Overburden management	Case 2*	Case 2*

*Case 2--Backfilling concurrent with mining - assumes 7 pits opened in 17-yr. mine life and the equivalent of one-pit overburden (2.4 yr. production) remains on the surface.

Ore Parameters (3.3.1.2)

Parameter	Average Large Mine	Average Mine
Average grade, percent U_3O_8	0.1	0.1
Th-232 concentration, pCi/g	10	10
Activity ratio (dust/ore)	2.5	2.5
Mineralogy	Sandstone	Sandstone
Density, MT/m^3	2.0	2.0
Surface area of stockpile, m^2	6,200	3,590
Area of pad, m^2	5,300	3,340
Stockpile height, m	9.2	3.1
Thickness of ore zone, m	12	12

Sub-Ore Parameters (3.3.1.3)

Parameter	Average Large Mine	Average Mine
Average grade, percent U_3O_8	0.015	0.015
Th-232 concentration, pCi/g	2	2
Activity ratio (dust/sub-ore)	2.5	2.5
Mineralogy	Sandstone	Sandstone
Density, MT/m^3	2.0	2.0
Surface area of stockpile, m^2	120,000	36,000
Stockpile height, m	30	30
Area of pad, hectares	11	3

Overburden Parameters (3.3.1.1)

Parameter	Average Large Mine	Average Mine
Average grade, percent U_3O_8	0.0020	0.0020
Th-232 concentration, pCi/g	1	1
Mineralogy	Sedimentary	Sedimentary
Density, MT/m^3	2.0	2.0
Surface area of dump, m^2	1.1×10^6	3.5×10^5
Dump height, m	65	30
Area of terrain, hectares	104	33

Wastewater Discharge Parameters (3.3.2.2)

Parameters (mg/ℓ except as noted)	Average Mine
Discharge volume, m^3/min	2.94 (Assumed value of 3.0)
Total uranium	0.07
Radium-226, pCi/ℓ (a)	0.41
Total suspended solids	20.88
Sulfate(b)	175
Zinc	0.071
Cadmium	0.004
Arsenic	0.005

(a) Concentration of Ra-226 and its daughters are reduced to 10% of the amount actually released due to irreversible sorption and precipitation.

(b) Concentration of sulfate is reduced to 20% of the amount actually released due to irreversible sorption and precipitation.

Airborne Source Terms (3.3.4)

Section 3.3.4 identifies and describes potential sources of airborne contamination at surface mines. The principal sources are dusts produced by mining operations and wind erosion and Rn-222 released by exposed uranium in the pit and overburden, sub-ore, and ore piles. The tables of Section 3.3.4 present the average annual emissions of contaminants from these sources during active mining.

Source	Table
Combustion Products	3.30
Vehicular Dusts	3.32
Dust from Mining Activities	3.33
Wind Suspended Dust	3.34
Rn-222 Emissions	3.35

4.2 Underground Mine

The model underground mine, defined in Section 3.4 as the "average large mine," will be located in New Mexico. However, to determine the total impact of all 305 underground uranium mines in the United States we used the parameters developed in Section 3.4 for the "average mine."

Production Parameters (1.3.1, 3.4.1)

Parameter	Average Large Mine	Average Mine
Ore, MT/yr	2×10^5	1.8×10^4
Sub-ore, MT/yr	2×10^5	1.8×10^4
Waste rock, MT/yr	2.2×10^4	2.0×10^3

Mining Parameters (3.4.1)

Parameter	Average Large Mine	Average Mine
Mining days per year	330	330
Mine life, yr	17	17
Ore stockpile residence time, days	41	41
Waste rock management	No backfill	No backfill

Ore Parameters (3.4.1.2)

Parameter	Average Large Mine	Average Mine
Average grade, percent U_3O_8	0.10	0.10
Th-232 concentration, pCi/g	10	10
Activity ratio (dust/ore)	2.5	2.5
Mineralogy	Sandstone	Sandstone
Density, MT/m^3	2.0	2.0
Surface area of stockpile, m^2	5,800	680
Stockpile height, m	3.1	3.1
Area of pad, m^2	5,480	620

Sub-Ore Parameters (3.4.1.3)

Parameter	Average Large Mine	Average Mine
Average grade, percent U_3O_8	0.035	0.035
Th-232 concentration, pCi/g	2	2
Activity ratio (dust/sub-ore)	2.5	2.5
Mineralogy	Sandstone	Sandstone
Density, MT/m^3	2.0	2.0
Surface area of dump, m^2	104,900	18,800
Dump height, m	12	6
Area of pad, m^2	99,400	17,700

Waste Rock Parameters (3.4.1.1)

Parameters	Average Large Mine	Average Mine
Average grade, percent U_3O_8	0.0020	0.0020
Th-232 concentration, pCi/g	1	1
Mineralogy	Sedimentary	Sedimentary
Density, MT/m^3	2.0	2.0
Surface area of dump, m^2	14,100	2,700
Dump height, m	12	6
Area of terrain, m^2	12,800	2,460

Wastewater Discharge Parameters (3.4.2.2)

Parameter (mg/ℓ except as noted)	Average Mine
Discharge volume, m^3/min	2.78 (assume value of 2.0)
Total Uranium	1.41
Radium-226, pCi/ℓ (a)	1.37
Lead-210, pCi/ℓ (a)	1.46
Total suspended solids	27.8
Sulfate (b)	116
Zinc	0.043
Barium	0.81
Cadmium	0.007
Arsenic	0.012
Molybdenum	0.29
Selenium	0.076

(a) Concentrations of Ra-226 and its daughters are reduced to 10 percent of the amount actually released due to irreversible sorption and precipitation.

(b) Concentrations of sulfate are reduced to 20 percent of the amount actually released due to irreversible sorption and precipitation.

Airborne Source Terms (3.4.4)

Section 3.4.4 identifies and describes potential sources of airborne contamination at underground mines. The principal sources are contaminated dusts due to mining operations and wind erosion and Rn-222 that is released from the mine exhaust vents during mining and from waste rock, sub-ore, and ore pile surfaces. Average annual emissions of contaminants from these sources during active mining operations are presented in the following tables of Section 3.4.4.

Source	Table
Combustion Products	3.52
Vehicular Dusts	3.56
Dust from Mining Activities	3.54
Wind Suspended Dust	3.55
Rn-222 Emissions	3.51

4.3 In Situ Leach Mine

The following parameters are for a model (hypothetical) in situ solution mine as defined in Section 3.5:

1. Size of deposit = 52.6 hectares
2. Average thickness of ore body = 8 m
3. Average ore grade = 0.06 percent U_3O_8
4. Mineralogy = Sandstone
5. Ore density = 2 MT/m^3
6. Ore body depth = 153 m
7. Mine life = 10 years (2-yr leach period in each of 5 sectors)
8. Well pattern = 5 spot
    Injection wells = 260
    Production wells = 200
    Monitoring wells = 80
9. Annual U_3O_8 production = 227 MT
10. Uranium leaching efficiency = 80 percent
11. Lixiviant = Alkaline
12. Lixiviant flow capacity = 2,000 ℓ/min
13. Lixiviant bleed = 50 ℓ/min (2.5 percent)
14. Uranium in Lixiviant = 183 mg/ℓ
15. Calcite ($CaCO_3$) removal required = 2 kg calcite per kg U_3O_8

Data were insufficient to estimate aqueous releases of contaminants from these type mines. However, since these facilities are planned to operate with no aqueous discharges, releases of contaminants via this pathway, except for possible excursions, should be small. Annual releases of contaminants to the atmosphere were computed in Section 3.5.3 for the model mine and listed in Table 3.59. These estimated annual airborne releases will be used to compute dose and indicate adverse health effects that might be associated with in situ leach mining.

4.4 Inactive Surface Mine

The model inactive surface mine will be located in Wyoming. It is defined in Section 3.7.1. The model mine parameters are listed below.

Mine Parameters

1. Period of active mining = 17 years
2. Total waste rock production = 8.88×10^5 MT
3. Total ore production = 3.59×10^4 MT
4. Density of ore and waste rock = 2.0 MT/m^3
5. Size of abandoned pit:
   Volume = 4.62×10^5 m^3
   Ground surface area = 2.03×10^4 m^2
   Pit bottom area = 6.00×10^3 m^2
   Depth = 36.7 m
6. Surface area and composition of waste rock pile = 6.33×10^4 m^2 uniformly covered to a depth of 0.36 m with sub-ore
7. Reclamation = none

Airborne Source Terms

Sections 3.7.1.1 and 3.7.1.2 identify and describe potential sources of airborne contamination at inactive surface uranium mines. The principal sources are contaminated, wind-suspended dust from the waste rock pile and Rn-222 released from exposed ore and sub-ore bearing surfaces in the pit and the waste rock pile. Tables 3.70 and 3.74 show average annual emissions of contaminants from these sources.

4.5 Inactive Underground Mine

The model inactive underground mine will be located in New Mexico. It is defined in Section 3.7.2, and its parameters are listed below.

Mine Parameters

1. Period of active mining = 15 yrs
2. Total waste rock production = 1.00×10^4 MT
3. Total ore production = 3.14×10^4 MT
4. Density of ore and waste rock = $2.0\ \text{MT/m}^3$
5. Surface area and composition of waste rock pile = $4.08 \times 10^3\ \text{m}^2$ uniformly covered to a depth of 0.78 m with sub-ore
6. Mine entrance and exhaust vents not sealed

Airborne Source Terms

Sections 3.7.2.1 and 3.7.2.2 identify and define potential sources of airborne contamination at inactive underground uranium mines. The principal sources are contaminated, wind-suspended dust from the waste rock pile and Rn-222 released from the unsealed mine entrance and exhaust vents and the waste rock pile. Tables 3.76 and 3.77 list average annual emission of contaminants from these sources.

SECTION 5

POTENTIAL PATHWAYS

5.0 Potential Pathways

5.1 General

5.1.1 Vegetation

Airborne particulate radioactivity may be deposited directly on the edible foliar surfaces of crops or on the soil and then migrate through the soil into the plant's root system and into an edible crop. Such crops may be consumed directly by man or by animals which are ultimately consumed by man. The use of contaminated water (either groundwater or surface) to irrigate crops may also lead to the ingestion of radionuclides from either the direct consumption of the crop or the crop-to-animal-to-man pathway.

The reconnaissance surveys of some inactive uranium mine sites indicated that no crops for human consumption were being farmed at or near any of the sites. Although the potential for man's ingestion of radionuclides in edible crops due to the direct deposition or the root uptake of either airborne particulates or contaminated mine water is a greater possibility near the active mines, farming in such areas is not extensive.

Almost every inactive and active mine site visited had range cattle and/or sheep grazing on the natural vegetation growing at the site; hence, the possible consumption of such animals could be a potential pathway for man's ingestion of radionuclides released into the environment surrounding the mine sites.

5.1.2 Wildlife

There are numerous species of mammals, birds, reptiles, and amphibians at both active and inactive uranium mine sites. Though mining may destroy their natural habitat, there are no significant radiological impacts on wildlife in these areas. Dewatering and drainage from active mines sometimes create ponds or streams that may be used by migratory waterfowl and local wildlife as a source of water, but, when mining is completed, the ponds dry up, probably without leaving any permanent or significant radiological impact on wildlife. The small lakes formed in inactive surface mine pits, however, may remain for a long period of time and have a significant environmental impact. It would be expected that sedimentation and eutrophication of the lakes would progressively diminish the impact with time by reducing the contact of ore bodies with the biosphere. The potential food pathway of animal-to-man via wildlife hunting at these sites is also minimal. Hunting is poor and hunting restrictions are usually observed at the mine sites.

5.1.3 Land Use

Most uranium mining activities have been conducted in areas away from population centers. Most mines are located on private property or are on Federal lands such as national forests. The predominant land use is as rangeland (or forest) and only minor areas are cropland. The fraction of land used for vegetable crop production for Wyoming and New Mexico is 1.59×10^{-3} and 1.38×10^{-3}, respectively. This fraction is based on the assumption that the statewide fractions apply to uranium mining areas within each state. Average population densities are typically rural, i.e., less than one person per 2.6 km^2.

5.1.4 Population Near Mining Areas

Uranium mines occur in clusters throughout many western states and are somewhat scattered throughout the eastern states. In order to estimate the number of persons residing within 50 miles (80km) of a mine, we used county populations where there either is or has been mining. Table 5.1 lists the states and their respective mining counties plus the numbers of inactive and active surface and underground uranium mines in each county. We derived the county population statistics from U.S. Department of Commerce census data (DOC78), which are January 1, 1975 estimates. The county areas were obtained from the same reference.

The area, 20,106 km^2, within a circle with a radius of 80 km usually exceeds the area of most counties. Because of this, the number of persons residing within 80 km of a mine will be underestimated using county population statistics. In other words, we consider the estimates of populations within the mining regions to be somewhat low.

Persons residing in a mining area are likely to be exposed from more than one mine because of the aforementioned clustering. To account for this, Table 5.1 lists the product (person-mines) for both active and inactive uranium mines. The total number of person-mines for inactive mines is approximately 82,000,000 persons. The total number of person-mines for active mines is approximately 14,000,000 persons. The combined equivalent population exposed to inactive and active uranium mining is approximately 96,000,000 persons.

Table 5.1 **Number of uranium mines and population statistics for counties containing uranium mines**

State	County	Number of Uranium Mines		Population Density	County Area	County Population	Person-Mines	Person-Mines
		Inactive	Active	(persons/km^2)	(km)^2	(persons)	Inactive	Active
Alaska	Southeast[a]	1	0	0.03	44,501	1,282	1,282	0
Arizona	Apache	140	0	1.1	28,930	32,304	4,522,560	0
	Cochise	2	0	3.8	16,203	61,918	123,836	0
	Coconino	113	0	1.0	48,019	48,326	5,460,838	0
	Gila	18	0	2.4	12,297	29,255	526,590	0
	Graham	1	0	1.4	11,961	16,578	16,578	0
	Maricopa	3	0	41.	23,711	971,228	2,913,684	0
	Mohave	5	0	0.76	34,232	25,857	129,285	0
	Navajo	35	1	2.3	25,666	59,649	2,088,715	59,649
	Pima	2	1	19.	23,931	443,958	887,916	443,958
	Santa Cruz	3	0	4.3	3,227	13,966	41,898	0
	Yavapai	3	0	1.8	20,956	37,005	111,015	0
California	Imperial	2	0	6.8	10,984	74,492	148,984	0
	Inyo	1	0	0.77	26,237	17,259	17,259	0
	Kern	6	0	17	21,113	349,874	2,099,244	0
	Lassen	2	0	1.4	11,816	16,796	33,592	0

Table 5.1 (Continued)

State	County	Number of Uranium Mines Inactive	Number of Uranium Mines Active	Population Density (persons/km^2)	County Area (km)^2	County Population (persons)	Person-Mines Inactive	Person-Mines Active
California	Madera	1	0	7.5	5,556	41,519	41,519	0
	Mono	1	0	0.51	7,840	4,016	4,016	0
	Riverside	5	0	25.	18,586	456,916	2,284,580	0
	San Bernardino	3	0	14	52,103	696,871	2,090,613	0
	Sierra	1	0	1.2	2,481	2,842	2,842	0
	Tuolumne	1	0	4.6	5,832	25,996	25,996	0
Colorado	Boulder	7	0	68	1,937	131,889	923,223	0
	Clear Creek	4	0	4.8	995	4,819	19,276	0
	Custer	3	0	0.59	1,909	1,120	3,360	0
	Dolores	6	0	0.62	2,657	1,641	9,846	0
	Eagle	2	0	1.7	4,353	7,498	14,996	0
	El Paso	1	0	42	5,587	235,972	235,972	0
	Fremont	25	0	6.6	4,022	26,545	663,625	0
	Garfield	10	0	2.3	7,759	17,845	178,450	0
	Gilpin	4	0	5.0	383	1,915	7,660	0
	Grand	4	0	0.86	4,802	4,107	16,428	0

Table 5.1 (Continued)

State	County	Number of Uranium Mines		Population Density (persons/km^2)	County Area (km)^2	County Population (persons)	Person-Mines Inactive	Person-Mines Active
		Inactive	Active					
Colorado	Gunnison	1	0	1.2	8,339	10,006	10,006	0
	Hinsdale	1	0	0.19	2,729	519	519	0
	Huerfano	2	0	1.6	4,077	6,590	13,180	0
	Jefferson	13	1	120	2,028	235,368	3,059,784	235,368
	La Plata	3	0	5.4	4,358	23,533	70,599	0
	Larimer	5	0	17	6,762	114,954	574,770	0
	Mesa	185	20	7.3	8,549	62,407	11,545,295	1,248,140
	Moffat	18	3	0.77	12,284	9,459	170,262	28,377
	Montezuma	6	1	2.7	5,423	14,642	87,852	14,642
	Montrose	479	63	3.5	5,796	20,286	9,716,994	1,278,018
	Park	7	0	0.77	5,599	4,311	30,177	0
	Pitkin	1	0	3.5	2,520	8,820	8,820	0
	Pueblo	1	0	20	6,228	124,560	124,560	0
	Rio Blanco	26	0	0.77	8,451	6,507	169,182	0
	Saguache	13	1	0.39	8,142	3,175	41,275	3,175
	San Juan	2	0	0.77	1,012	779	1,558	0

Table 5.1 (Continued)

State	County	Number of Uranium Mines Inactive	Number of Uranium Mines Active	Population Density $(persons/km^2)$	County Area $(km)^2$	County Population (persons)	Person-Mines Inactive	Person-Mines Active
Colorado	San Miguel	339	25	0.77	3,322	2,557	866,823	63,925
	Teller	3	0	3.9	1,432	5,584	16,752	0
Idaho	Custer	5	0	0.23	12,766	2,967	14,835	0
	Lemhi	1	0	0.39	11,862	6,395	6,395	0
Montana	Broadwater	1	0	0.82	3,090	2,526	2,526	0
	Carbon	11	0	1.5	5,325	7,797	85,767	0
	Fallon	1	0	0.96	4,229	4,050	4,050	0
	Hill	1	0	2.3	7,581	17,358	17,358	0
	Jefferson	3	0	1.5	4,278	6,839	20,517	0
	Madison	1	0	0.55	9,138	5,014	5,014	0
Nevada	Clark	2	0	16.2	20,393	330,714	661,428	0
	Elko	3	0	0.31	44,452	13,958	41,874	0
	Humboldt	1	0	0.25	25,128	6,375	6,375	0
	Lander	2	0	0.39	14,558	2,992	5,984	0
	Lincoln	2	0	0.19	27,114	2,647	5,294	0
	Lyon	2	0	1.9	5,257	10,508	21,016	0

Table 5.1 (Continued)

State	County	Number of Uranium Mines Inactive	Number of Uranium Mines Active	Population Density (persons/km^2)	County Area (km)^2	County Population (persons)	Person-Mines Inactive	Person-Mines Active
Nevada	Mineral	2	0	0.71	9,751	7,051	14,102	0
	Nye	1	0	0.12	46,786	5,599	5,599	0
	Washoe	6	0	8.9	16,487	144,750	868,500	0
New Jersey	Sussex	1	0	73	1,364	99,299	99,299	0
New Mexico	Catron	4	0	0.12	17,863	2,198	8,792	0
	Dona Ana	1	0	7.1	9,852	69,773	69,773	0
	Grant	3	0	2.1	10,282	22,030	66,090	0
	Harding	1	0	0.25	5,527	1,348	1,348	0
	Hidalgo	1	0	0.53	8,927	4,734	4,734	
	McKinley	73	35	3.5	14,138	49,483	3,612,259	1,731,905
	Mora	1	0	0.93	5,025	4,673	4,673	0
	Quay	3	0	1.5	7,446	10,903	32,709	0
	Rio Arriba	8	0	1.9	15,133	28,752	230,016	0
	Sandoval	3	0	2.3	9,619	22,123	66,369	0
	San Juan	41	0	4.6	14,245	65,527	2,686,607	0
	San Miguel	3	0	1.8	12,279	21,951	65,853	0
	Santa Fe	2	0	13	4,926	64,038	128,076	0

Table 5.1 (Continued)

State	County	Number of Uranium Mines Inactive	Number of Uranium Mines Active	Population Density (persons/km^2)	County Area (km)^2	County Population (persons)	Person-Mines Inactive	Person-Mines Active
New Mexico	Sierra	6	0	0.67	10,790	7,189	43,134	0
	Socorro	7	0	0.57	17,102	9,763	68,341	0
	Taos	1	0	3.0	5,843	17,516	17,516	0
	Valencia	19	4	3.1	14,649	45,411	862,809	181,644
North Dakota	Billings	9	0	0.39	2,950	1,153	10,377	0
	Slope	1	0	0.39	3,172	1,360	1,360	0
	Stark	3	0	5.8	3,408	19,650	58,950	0
Oklahoma	Caddo	2	0	8.8	3,294	28,931	57,862	0
	Custer	1	0	8.3	2,538	21,040	21,040	0
Oregon	Crook	1	0	1.3	7,705	9,985	9,985	0
	Lake	2	0	0.34	21,318	7,158	14,316	0
South Dakota	Butte	3	0	1.3	5,827	7,825	23,475	0
	Custer	10	0	1.2	4,032	5,196	51,960	0
	Fall River	93	0	1.9	4,514	8,066	750,138	0
	Harding	28	0	0.39	6,946	1,879	52,612	0
	Lawrence	2	0	8.4	2,072	17,453	34,906	0
	Pennington	5	0	8.3	7,198	59,349	296,745	0

Table 5.1 (Continued)

State	County	Number of Uranium Mines Inactive	Number of Uranium Mines Active	Population Density (persons/km^2)	County Area (km)^2	County Population (persons)	Person-Mines Inactive	Person-Mines Active
Texas	Briscoe	2	0	1.2	2,264	2,794	5,588	0
	Burnet	1	0	4.4	2,577	11,420	11,420	0
	Crosby	1	0	3.9	2,359	9,085	9,085	0
	Garza	6	0	2.8	2,367	6,611	39,666	0
	Gonzales	2	0	5.8	2,735	16,342	32,684	0
	Karnes	23	10	6.6	1,963	12,955	297,965	129,550
	Live Oak	6	5	2.3	2,732	6,453	38,718	32,265
Utah	Beaver	9	1	0.77	6,692	5,152	46,368	5,152
	Box Elder	1	0	1.9	14,512	28,129	28,129	0
	Duchesne	4	0	1.5	8,430	12,645	50,580	0
	Emery	186	18	0.39	11,497	4,483	833,838	80,694
	Garfield	131	15	0.39	13,359	5,210	682,510	78,150
	Grand	164	17	0.77	9,536	7,342	1,204,088	124,814
	Iron	1	0	1.4	8,547	12,177	12,177	0
	Juab	4	0	0.52	8,837	4,574	18,296	0
	Kane	3	0	0.39	10,111	3,943	11,829	0

Table 5.1 (Continued)

State	County	Number of Uranium Mines		Population Density (persons/km^2)	County Area (km)^2	County Population (persons)	Person-Mines Inactive	Person-Mines Active
		Inactive	Active					
Utah	Piute	10	0	0.77	1,952	1,503	15,030	0
	San Juan	241	24	0.77	19,961	15,369	3,703,929	368,856
	Sevier	2	0	2.3	4,996	11,490	22,980	0
	Uintah	14	0	1.5	11,621	17,431	244,034	0
	Washington	6	0	2.7	6,285	16,969	101,814	0
	Wayne	32	0	0.39	6,438	2,510	80,320	0
Washington	Pend Oreille	3	0	1.9	3,631	7,361	22,083	0
	Spokane	9	0	67	4,553	306,338	2,757,042	0
	Stevens	1	2	3.5	6,425	22,489	22,489	44,978
Wyoming	Albany	4	0	2.3	11,002	25,304	101,216	0
	Big Horn	9	0	1.5	8,176	12,264	110,376	0
	Campbell	55	0	1.2	12,318	14,781	812,955	0
	Carbon	16	3	0.77	20,473	15,764	252,224	47,292
	Converse	31	5	0.77	11,087	8,536	264,616	42,680
	Crook	23	0	0.77	7,464	5,747	132,181	0
	Fremont	65	13	1.2	23,817	28,580	1,857,700	371,540

Table 5.1 (Continued)

State	County	Number of Uranium Mines Inactive	Number of Uranium Mines Active	Population Density (persons/km^2)	County Area (km)^2	County Population (persons)	Person-Mines Inactive	Person-Mines Active
Wyoming	Johnson	15	0	0.39	10,813	4,217	63,255	0
	Natrona	16	2	3.9	13,835	53,956	863,296	107,912
	Niobrara	13	0	0.39	6,770	2,640	34,320	0
	Sublette	1	0	0.39	12,564	4,899	4,899	0
	Sweetwater	4	2	1.2	27,011	32,413	129,652	64,826
	Washakie	2	0	1.5	5,858	8,787	17,574	0
	Weston	1	0	1.0	6,234	6,307	6,307	0
				Average Population Density	Total County Area (km)^2	Total County Population	Total Person-Mines (Inactive)	Total Person-Mines (Active)
				4.4 persons/km^2	1,492,136	6,625,099	82,327,885	14,035,161

Note.--Population statistics from (DOC78).

[a] Congressional District.

5.1.5 Population Statistics of Humans and Beef Cattle

Table 5.2 lists some population statistics for humans in New Mexico and Wyoming, humans in all uranium mining states, and beef cattle in New Mexico and Wyoming.

Table 5.2 Population statistics for humans and beef cattle

	New Mexico	Wyoming	All Uranium Mining States
Total Human and Beef Cattle Population Within 80 km Radius of Mines			
Human	447,412	224,195	6,625,099
Beef cattle	753,000	905,000	----------
Average Human and Beef Cattle Population Densities Within 80 km Radius of Uranium Mines (number/km^2)[a]			
Human	2.4	1.3	4.4
Beef Cattle	4.1	5.1	----

[a] Areas taken from Table 5.1: New Mexico = 183,646 km^2; Wyoming = 177,422 km^2, and the total county area = 1,492,136 km^2.

5.2 Prominent Environmental Pathways and Parameters for Aqueous Releases

From a computer code prepared within EPA, we calculated annual committed dose equivalents to individuals and annual collective dose equivalents to a population for these assessments. Table 5.3 lists the aqueous pathways that were initially considered potential pathways of exposure. As indicated in Table 5.3, these pathways result in computation of dose equivalents due to inhalation, ingestion, ground surface exposure, and air submersion. For above surface crop ingestion, milk ingestion, and beef ingestion (pathways 3, 4, and 5), we considered only uptake through the plant root systems to predict concentrations of radionuclides in crops, since essentially all irrigation is ditch irrigation. Appendix J contains a detailed explanation of the environmental transport and dosimetry models used in these analyses.

The maximum individual for the aquatic pathways is the individual at maximum risk. He is exposed to radionuclides discharged in mine effluent through pathways 2 through 10 of Table 5.3. The water contributing radionuclides to these pathways comes from a creek into which a mine discharges. The average individual is exposed to the average risk of all persons included in the population of the assessment area. He is exposed to radionuclides discharged in mine effluent through pathways 2 through 8 and 10 of Table 5.3. The water contributing radionuclides to these pathways is taken from the regional river after the creek water has been diluted in this river. The population considered in the assessment of the aquatic pathways is obtained by multiplying the regional assessment area size by the population density within this area. This assessment area contains the drainage basin for the mine effluent stream, the creek and the regional river discussed in defining the maximum and average individuals.

5.2.1 Individual Committed Dose Equivalent Assessment

Section 6 of this report contains the computed dose equivalents to the maximum individual and to the average individual. For the maximum individual, we included all pathways in Table 5.3 except drinking water (pathway 1). It is known that the releases to the aquatic environment occur through discharge of mine water to surface streams. Potentially, drinking water could be one of the most significant pathways for the maximum individual dose equivalents, if surface water containing mine wastes was drunk. However, it appears that all drinking water for both the New Mexico and the Wyoming sites comes from wells (Robert Kaufmann, 1979, U.S. Environmental Protection Agency, Las Vegas, NV, personal communication). Thus, the only way mine discharges can enter human drinking water is by percolating through the soil. Since we do not know the soil chemistry for these sites well enough to predict the ion-exchange parameters for the soil, we can not predict, realistically, the quantity of mine-related radionuclides that would reach the groundwater. We expect that these ion-exchange factors would be large for several of the radionuclides considered in these analyses and that groundwater concentrations of radionuclides discharged in mine water would be quite small compared to concentrations in the surface water downstream from the mines. Further study is needed before dose equivalents for the maximum individual by drinking groundwater can be adequately addressed.

The following are other assumptions used to calculate maximum individual dose equivalents:

1. Ground surface concentrations of radionuclides (used for pathways 6 through 8) are for 8.5 years, the assumed midpoint of mine life. The assumed period of mine operation is 17 years. The organ annual dose equivalents for the external surface exposure pathway are based on the ground concentrations after the 8.5 years buildup time.
2. For inhaled or ingested radionuclides, the dose equivalents are the annual committed dose equivalents that will be accumulated over 70 years after intake for an adult.

We calculated dose equivalents to the average individual in the assessment area by taking the population dose equivalents (discussed in Subsection 5.2.2) and dividing by the population living in the area.

Table 5.3 Aquatic environmental transport pathways initially considered

Pathway No.	Pathway
1	Drinking water ingestion
2	Freshwater fish ingestion
3	Above surface crops ingestion - irrigated cropland
4	Milk ingestion - cows grazing on irrigated pasture
5	Beef ingestion - cows grazing on irrigated pasture
6	Inhalation - material resuspended which was deposited during irrigation
7	External dose due to ground contamination by material originally deposited during irrigation
8	External dose due to air submersion in resuspended material originally deposited during irrigation
9	Milk ingestion - cows drinking contaminated surface water
10	Beef ingestion - cows drinking contaminated surface water

5.2.2 Collective (Population) Dose Equivalent Assessment

For the population dose equivalent assessment calculations, we concluded that the pathways of concern are pathways 2, 3, 4, 5, 6, 7, 8, and 10 of Table 5.3 (detailed discussion in Appendix J, subsection J2). The size of the assessment areas for New Mexico is 19,037 km^2 and 13,650 km^2 for Wyoming. We used the following considerations to calculate population dose equivalents for the assessment area:

1. Ground surface concentrations of radionuclides are for 8.5 years, the assumed midpoint mine life. (The period of mine operation is 17 years.) The organ annual collective dose equivalent rates for the external surface exposure pathway are based on the ground concentrations after the 8.5 year buildup time.

2. For inhaled or ingested radionuclides, the dose equivalents are the annual collective dose equivalents that will be accumulated over the 70 years after intake for adults.

3. The population distributions around the sites are based on estimates by county planners (John Zaboroc, 1979, Converse Area Planning Office, Douglas, Wyoming, personal communication) and agricultural personnel (Tony Romo, 1979, Valencia County Agent, Los Lunas, New Mexico, personal communication) for 1979. The populations, assumed to remain constant in time, were estimated to be 16,230 and 64,950 persons in the Wyoming and New Mexico assessment areas, respectively.

4. Average agricultural production data for the county which contains a major portion of the assessment area are used.

5. The population in the assessment area eats food from the assessment area. We assume that any imported food is free of radionuclides.

As mentioned previously, Appendix J contains the details regarding the models and values for parameters used in these analyses.

5.3 Prominent Environmental Pathways and Parameters for Atmospheric Releases

We used the AIRDOS-EPA (Mo79) computer code to calculate radionuclide air and ground concentrations, ingestion and inhalation intakes, and working level exposures; and we used the DARTAB (Be80) computer code to calculate dose and risk from the AIRDOS-EPA intermediate output using dose and risk factors from the RADRISK (Du80) computer code. We calculated working levels associated with Rn-222 emissions assuming that Rn-222 decay products were 70 percent in equilibrium with Rn-222, a value considered representative of indoor exposure conditions (Ge78). Appendix K contains a detailed discussion of the application of the AIRDOS-EPA and RADRISK computer codes.

Figure 5.1 shows the general airborne pathways evaluated for uranium mines. We calculated doses due to air immersion, ground surface exposure, inhalation, and ingestion of radionuclides, but we did not address the resuspension pathway, since the AIRDOS-EPA code did not provide a method for calculating resuspended air concentrations or subsequent redeposition to the ground surface. We used the modification to the AIRDOS-EPA computer code made by Nelson (Ne80) to include the effect of environmental removal of radioactivity from the soil. For ingestion, transfers associated with both root uptake and foliar deposition on food and forage are considered.

5.3.1 Individual Committed Dose Equivalent Assessment

We assessed the maximum individual on the following basis:

1. The maximum individual for each source category is intended to represent an average of the individuals living close to each model uranium mine. The individual is assumed to be located about 1600 meters from the center of the model site.
2. Ground surface concentrations of radionuclides used in the assessment are those that would occur during the midpoint of the active life of the model uranium mine. Buildup times used in the assessment are 8.5 years for active surface and underground mines, 5 years for the in situ leach mine, and 26.5 years for the inactive surface and underground mines. The 26.5-year buildup time for the inactive mines is chosen to represent the midpoint of the 53-year exposure time that a resident living a lifetime in the region around the model mine is estimated to experience. The organ dose equivalent rates for the external surface exposure pathway are based on

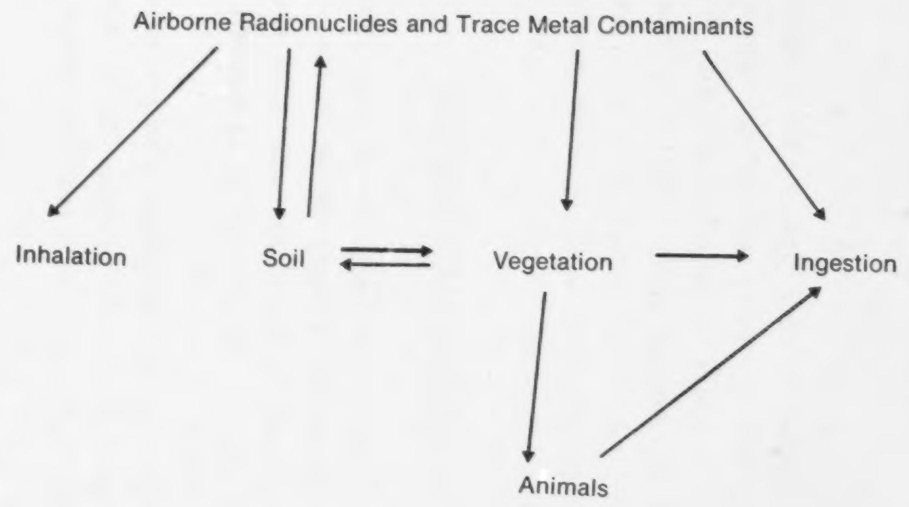

Figure 5.1 Potential airborne pathways in the vicinity of uranium mines.

the concentrations for the indicated buildup time.

3. For inhaled or ingested radionuclides, the dose equivalent rates are actually the 70-year committed dose equivalent rates for an adult receptor, i.e., the internal dose equivalent that would be delivered up to 70 years after an intake. The individual dose equivalent rates in the tables are in units of mrem/yr.

4. The individual is assumed to home grow a portion of his or her diet consistent with the rural setting for each model uranium mine site. Appendix K contains the actual fractions of home-produced food consumed by individuals for the model mine sites. The portion of the individual's diet that was not locally produced is assumed to be imported and uncontaminated by the assessment source.

5.3.2 Collective (Population) Dose Equivalent Assessment

The collective dose equivalent assessment to the population out to 80 km from the facility under consideration is performed as follows:

1. The population distribution around the model mine sites is based on the 1970 census. The population is assumed to remain constant in time.
2. Ground surface concentrations and organ dose equivalent rates for the external surface exposure pathway (as for the individual case) are those that would occur over the active life of the model mine.
3. Average agricultural production data for the state in which the model uranium mine is located are assumed.
4. The population in the assessment area eats food from the assessment area to the extent that the calculated production allows, and any balance is assumed to be imported without contamination by the assessment source.
5. Seventy-year committed dose equivalent factors for an adult receptor (as for the individual case) are used for ingestion and inhalation.

5.4 Mine Wastes Used In the Construction of Habitable Structures

Using uranium mine wastes under or around habitable structures or building habitable structures on land contaminated with uranium mine wastes can result in increased radiation exposures to individuals occupying these structures. The radium-226 present in these wastes elevates the concentrations of radon-222 and its decay products and produces increased gamma radiation inside these structures. The health risk to individuals occupying these structures is generally much greater from inhaling radon-222 decay products than the risk received from gamma radiation.

Radon-222, formed from the decay of radium-226, is an inert gas that diffuses through the soil and migrates readily through foundations, floors, and walls and accumulates in the inside air of a structure. Breathing radon-222 and its short-lived decay products (principally polonium-218, bismuth-214, and polonium-214) exposes the lungs to radiation.

The radon-222 decay product concentration (working level) inside a structure from radon-222 gas diffusing from underlying soil is extremely variable and influenced by many complex factors. These would include the radium-226 concentration of the soil, the fraction of radon-222 emanating from the soil, the diffusion coefficient of radon-222 in soil, the rate of influx of radon-222 into the structure, the ventilation rate of the structure, and the amount of plate-out (adsorption) of radon-222 decay products on inside surfaces.

The potential risks of fatal lung cancer that could occur to individuals living in homes built on land contaminated by uranium mine wastes have been estimated using measurements and calculational methodology relating radon-222 decay product concentrations inside homes to the radium-226 concentrations in outside soil (He78, Wi78). These estimates are shown in Section 6.1.5.

5.5 References

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SECTION 6

HEALTH AND ENVIRONMENTAL EFFECTS

6.0 Health and Environmental Effects

6.1 Health Effects and Radiation Dosimetry

6.1.1 Radioactive Airborne Emissions

We used data on radioactive emissions (Section 3) to estimate the public health impact of these emissions. Our assessments include estimates of the following radiation exposures and health risks:

1. Dose equivalent rates and working level exposures to the most exposed individuals (maximum individual) and to the average exposed individuals in the regional population (average individual)
2. Collective dose equivalent rates and working level exposures to the regional population
3. Lifetime fatal cancer risks to the maximum and average individuals in the regional population
4. Genetic effect risk to the descendants of the maximum and average individuals in the regional population
5. The number of fatal cancers committed in the regional population per year of model mine operation
6. The number of genetic effects committed to the descendants of the regional population per year of model mine operation

The somatic health impact risks estimated in this report are for fatal cancers only. For whole body exposure, the risk of nonfatal cancer is about the same or slightly less than for fatal cancer. Thus, for whole body doses, it is conservatively estimated that one nonfatal cancer could occur for each additional fatal cancer. The somatic health impact for the regional population (additional cancers per year) is calculated at equilibrium for continuous exposure and this is equal to the additional cancers committed over all time per year of exposure; thus we used the term committed additional cancers (see Appendix L).

The genetic effect risks estimated in this report are for effects in descendants of an irradiated parent or parents. Genetic effects per year in the regional population due to radionuclide releases from the mines are calculated for an equilibrium exposure situation. The calculated genetic effects per year at equilibrium is equal to the genetic effects committed over all time from one year exposure. Thus, the calculated additional

Table 6.1 Annual release rates (Ci) used in the dose equivalent and health effects computations for active uranium mines

Classification	Location	Average Surface Mine[a]			Average Large Surface Mine [a]		
		U	Th	Rn-222	U	Th	Rn-222
Mining activities	Pit/mine site	4.3E-3	2.2E-4	1.99E+2	2.57E-2	1.44E-3	7.97E+2
Ore	Pile site	1.01E-2	1.42E-4	4.2E+1	4.42E-2	6.20E-4	9.6E+1
Sub-ore	Pile site	4.2E-4	8.4E-6	5.0E+1	1.51E-3	3.00E-5	1.66E+2
Overburden/ waste rock	Pile site	2.25E-3	1.50E-4	4.0E+1	1.34E-2	8.94E-4	2.02E+2
Vehicular dust	Mining area	9.9E-4	3.7E-4	0	5.86E-3	2.17E-3	0
Total	All sources	1.81E-2	8.90E-4	3.31E+2	9.07E-2	5.15E-3	1.26E+3

(a) Release rates taken from Tables 3.32 to 3.35.
(b) Release rates taken from Tables 3.51 and 3.54 to 3.56.

Table 6.1 (cont.)

Average Underground Mine [b]			Average Large Underground Mine [b]			In Situ Leach Mine [c]		
U	Th	Rn-222	U	Th	Rn-222	U	Th	Rn-222
2.22E-4	2.8E-6	3.08E+2	2.41E-3	3.10E-5	3.42E+3	1.0E-1	0	6.50E+2
9.63E-4	1.35E-5	7.7	1.07E-2	1.50E-4	6.83E+1	N.A. [d]	N.A.	N.A.
1.04E-3	8.4E-6	6.1E+1	5.95E-3	4.8E-5	3.38E+2	N.A.	N.A.	N.A.
9.6E-6	6.4E-7	5.0E-1	5.10E-5	3.40E-6	2.6	N.A.	N.A.	N.A.
6.5E-5	2.4E-5	0	1.29E-4	4.80E-5	0	N.A.	N.A.	N.A.
2.30E-3	4.93E-5	3.77E+2	1.92E-2	2.80E-4	3.83E+3	1.0E-1	0	6.50E+2

[c] Release rates taken from Table 3.59.

[d] N.A.- Not Applicable.

Note.--Columns labeled U and Th include each daughter of the decay chain in secular equilibrium.

Table 6.2 Annual release rates (Ci) used in the dose equivalent and health effects computations for inactive uranium mines

	Surface Mine (a)			Underground Mine (b)		
Location	U	Th	Rn-222	U	Th	Rn-222
Pit/vents-portals	0	0	8.1	0	0	7.55
Waste rock/ sub-ore pile	1.48E-3	1.1E-5	1.74E+1	2.38E-4	1.7E-6	1.7

(a) Release rates taken from Tables 3.70 and 3.74.

(b) Release rates taken from Tables 3.76 and 3.77.

Note.--Column headings U and Th include each daughter of the decay chain in secular equilibrium.

genetic effects are committed effects to all future generations for one year of exposure to the regional population.

We calculated individually each major source of radionuclide airborne emissions for each model uranium mine site so that we could determine the extent that each source contributed to the total health impact. Tables 6.1 and 6.2 contain the annual release rates for each source classification (or location) that we used to calculate dose equivalent rates and health effects for active and inactive uranium mines.

The estimated annual working level exposures from Rn-222 emissions by the model uranium mines are listed in Table 6.3. The working level exposures presented for the maximum individual are the Rn-222 decay product levels to which an individual would be continuously exposed for an entire year. Working level exposure to the regional population is the sum of the exposures to all individuals in the exposed population from the annual release from the model mine.

We estimated radiological impacts of radioactive airborne emissions from the model uranium mines with the AIRDOS-EPA (Mo79), RADRISK (Du80), and DARTAB (Be80) computer codes. Appendixes K and L contain explanations of our use of these computer codes.

Where emissions for U-238 plus daughters and Th-232 plus daughters were reported (Section 3), a source term for both the parent and important daughters were input into the AIRDOS-EPA code. For example, a reported emission rate of 0.01 Ci/yr of U-238 plus daughters (U in Tables 6.1 and 6.2) would be input into the AIRDOS-EPA code as 0.01 Ci/yr of U-238, 0.01 Ci/yr of U-234, 0.01 Ci/yr of Th-230, 0.01 Ci/yr of Ra-226, 0.01 Ci/yr of Pb-214, 0.01 Ci/yr of Bi-214, 0.01 Ci/yr of Pb-210, and 0.01 Ci/yr of Po-210. A reported emission rate of 0.01 Ci/yr of Th-232 plus daughters (Th in Tables 6.1 and 6.2) would be input into the AIRDOS-EPA code as 0.01 Ci/yr of Th-232, 0.01 Ci/yr of Ra-228, 0.01 Ci/yr of Ac-228, 0.01 Ci/yr of Th-228, 0.01 Ci/yr of Ra-224, 0.01 Ci/yr of Pb-212, 0.01 Ci/yr of Bi-212, and 0.0036 Ci/yr of Tl-208. The Tl-208 source term is approximately one-third that of Bi-212 because of the branching ratio.

The maximum individual, average individual, and population dose equiv-

Table 6.3 Annual working level exposure from radon-222 emissions from model uranium mines

Source	Maximum Individual (WL)[a]	Average Individual (WL)	Regional Population (person-WL)
Average Surface Mine	2.3E-4	4.5E-7	6.5E-3
Average Large Surface Mine	8.4E-4	1.7E-6	2.5E-2
Average Underground Mine	4.6E-4	2.1E-6	7.5E-2
Average Large Underground Mine	4.7E-3	2.1E-5	7.6E-1
Inactive Surface Mine	1.8E-5	3.5E-8	5.0E-4
Inactive Underground Mine	1.1E-5	5.1E-8	1.8E-3
In Situ Leach Mine	4.5E-4	8.9E-7	1.3E-2

(a) Working level.

Table 6.4 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model average surface uranium mine

Organ	Maximum Individual (mrem/yr)	Average Individual (mrem/yr)	Population (person-rem/yr)
Red marrow	2.4	5.4E-3	7.7E-2
Endosteal	3.4E+1	7.5E-2	1.1
Pulmonary	1.2E+1	6.3E-3	9.0E-2
Muscle	5.5E-1	2.0E-3	2.7E-2
Liver	1.6	6.3E-3	9.1E-2
Stomach wall	9.7E-2	8.9E-5	1.3E-3
Pancreas	5.2E-1	1.9E-3	2.7E-2
LLI(a) wall	4.6E-1	1.6E-3	2.3E-2
Kidney	4.2	1.8E-2	2.5E-1
Bladder wall	3.0E-1	9.7E-4	1.4E-2
ULI(b) wall	2.1E-1	5.2E-4	7.4E-3
SI(c) wall	9.4E-2	1.2E-4	1.7E-3
Ovaries	5.1E-1	1.9E-3	2.7E-2
Testes	5.4E-1	1.9E-3	2.7E-2
Spleen	6.4	2.8E-2	4.0E-1
Uterus	5.1E-1	1.9E-3	2.7E-2
Thymus	5.2E-1	1.9E-3	2.7E-2
Thyroid	5.4E-1	1.9E-3	2.7E-2
Weighted mean	4.9	5.5E-3	7.8E-2

(a) Lower large intestine wall.
(b) Upper large intestine wall.
(c) Small intestine wall.

Table 6.5 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model average large surface uranium mine

Organ	Maximum Individual (mrem/yr)	Average Individual (mrem/yr)	Population (person-rem/yr)
Red marrow	1.35E+1	2.7E-2	3.9E-1
Endosteal	1.9E+2	3.8E-1	5.4
Pulmonary	6.6E+1	3.1E-2	4.5E-1
Muscle	3.0	9.6E-3	1.4E-1
Liver	8.9	3.2E-2	4.6E-1
Stomach wall	5.4E-1	4.5E-4	6.4E-3
Pancreas	3.0	9.6E-3	1.4E-1
LLI wall	2.5	8.2E-3	1.2E-1
Kidney	2.1E+1	9.0E-2	1.3
Bladder wall	1.7	4.9E-3	7.0E-2
ULI wall	1.1	2.6E-3	3.8E-2
SI wall	5.2E-1	6.0E-4	8.6E-3
Ovaries	2.8	9.6E-3	1.4E-1
Testes	3.0	9.6E-3	1.4E-1
Spleen	3.5E+1	1.4E-1	2.0
Uterus	2.8	9.6E-3	1.4E-1
Thymus	2.9	9.6E-3	1.4E-1
Thyroid	3.0	9.6E-3	1.4E-1
Weighted mean	2.7E+1	2.7E-2	3.8E-1

Table 6.6 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model average underground uranium mine

Organ	Maximum Individual (mrem/yr)	Average Individual (mrem/yr)	Population (person-rem/yr)
Red marrow	5.1E-1	8.3E-4	2.9E-2
Endosteal	7.2	1.2E-2	4.1E-1
Pulmonary	2.9	5.0E-3	1.8E-1
Muscle	1.2E-1	2.3E-4	8.3E-3
Liver	3.5E-1	7.2E-4	2.7E-2
Stomach wall	2.0E-2	2.8E-5	1.0E-3
Pancreas	1.1E-1	2.2E-4	8.0E-3
LLI wall	9.4E-2	1.8E-4	6.5E-3
Kidney	9.1E-1	2.0E-3	7.4E-2
Bladder wall	6.4E-2	1.2E-4	4.4E-3
ULI wall	4.3E-2	7.3E-5	2.7E-3
SI wall	2.0E-2	2.8E-5	1.0E-3
Ovaries	1.1E-1	2.2E-4	8.0E-3
Testes	1.1E-1	2.3E-4	8.0E-3
Spleen	1.4	3.1E-3	1.1E-1
Uterus	1.1E-1	2.2E-4	7.9E-3
Thymus	1.1E-1	2.2E-4	8.0E-3
Thyroid	1.1E-1	2.3E-4	8.1E-3
Weighted mean	1.1	2.0E-3	7.1E-2

Table 6.7 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model average large underground uranium mine

Organ	Maximum Individual (mrem/yr)	Average Individual (mrem/yr)	Population (person-rem/yr)
Red marrow	4.2	6.9E-3	2.5E-1
Endosteal	6.0E+1	9.6E-2	3.5
Pulmonary	2.5E+1	4.7E-2	1.7
Muscle	9.7E-1	1.9E-3	6.9E-2
Liver	2.9	6.0E-3	2.2E-1
Stomach wall	1.7E-1	2.3E-4	8.5E-3
Pancreas	9.4E-1	1.8E-3	6.8E-2
LLI wall	7.8E-1	1.5E-3	5.5E-2
Kidney	7.7	1.7E-2	6.2E-1
Bladder wall	5.4E-1	1.0E-3	3.6E-2
ULI wall	3.6E-1	6.0E-4	2.2E-2
SI wall	1.6E-1	2.3E-4	8.4E-3
Ovaries	9.2E-1	1.8E-3	6.6E-2
Testes	9.4E-1	1.8E-3	6.8E-2
Spleen	1.2E+1	2.6E-2	9.2E-1
Uterus	9.2E-1	1.8E-3	6.6E-2
Thymus	9.4E-1	1.8E-3	6.7E-2
Thyroid	9.4E-1	1.9E-3	6.8E-2
Weighted mean	9.8	1.8E-2	6.2E-1

Table 6.8 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model inactive surface uranium mine

Organ	Maximum Individual (mrem/yr)	Average Individual (mrem/yr)	Population (person-rem/yr)
Red marrow	2.1E-1	4.8E-4	6.9E-3
Endosteal	2.9	6.8E-3	9.8E-2
Pulmonary	9.5E-1	5.0E-4	7.2E-3
Muscle	5.6E-2	1.8E-4	2.6E-3
Liver	1.4E-1	5.5E-4	7.8E-3
Stomach wall	1.5E-2	1.1E-5	1.6E-4
Pancreas	5.4E-2	1.8E-4	2.6E-3
LLI wall	4.4E-2	1.4E-4	2.0E-3
Kidney	3.5E-1	1.5E-3	2.1E-2
Bladder wall	3.3E-2	9.2E-5	1.3E-3
ULI wall	2.4E-2	4.7E-5	6.7E-4
SI wall	1.4E-2	1.3E-5	1.8E-4
Ovaries	5.2E-2	1.8E-4	2.5E-3
Testes	5.5E-2	1.8E-4	2.6E-3
Spleen	5.3E-1	2.3E-3	3.3E-2
Uterus	5.2E-2	1.8E-4	2.5E-3
Thymus	5.3E-2	1.8E-4	2.5E-3
Thyroid	5.5E-2	1.8E-4	2.6E-3
Weighted mean	3.9E-1	4.7E-4	6.8E-3

Table 6.9 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a model inactive underground uranium mine

Organ	Maximum Individual (mrem/yr)	Average Individual (mrem/yr)	Population (person-rem/yr)
Red marrow	5.8E-2	9.3E-5	3.4E-3
Endosteal	8.0E-1	1.3E-3	4.6E-2
Pulmonary	2.7E-1	3.4E-4	1.3E-2
Muscle	1.6E-2	2.9E-5	1.0E-3
Liver	3.9E-2	7.9E-5	2.8E-3
Stomach wall	4.0E-3	5.2E-6	1.8E-4
Pancreas	1.5E-2	2.8E-5	1.0E-3
LLI wall	1.2E-2	2.2E-5	8.0E-4
Kidney	9.7E-2	2.1E-4	7.6E-3
Bladder wall	9.1E-3	1.6E-5	5.8E-4
ULI wall	6.6E-3	1.0E-5	3.7E-4
SI wall	3.7E-3	4.9E-6	1.8E-4
Ovaries	1.4E-2	2.7E-5	9.7E-4
Testes	1.5E-2	2.8E-5	1.0E-3
Spleen	1.5E-1	3.2E-4	1.2E-2
Uterus	1.4E-2	2.7E-5	9.8E-4
Thymus	1.5E-2	2.8E-5	1.0E-3
Thyroid	1.5E-2	2.8E-5	1.0E-3
Weighted mean	1.1E-1	1.5E-4	5.7E-3

Table 6.10 Annual radiation dose equivalents due to atmospheric radioactive particulate and Rn-222 emissions from a hypothetical in situ uranium solution mine

Organ	Maximum Individual (mrem/yr)	Average Individual (mrem/yr)	Population (person-rem/yr)
Red marrow	1.6E-1	2.7E-4	3.8E-3
Endosteal	2.8	5.0E-3	7.1E-2
Pulmonary	3.9E+1	2.0E-2	2.9E-1
Muscle	8.4E-3	2.2E-5	3.1E-4
Liver	1.9E-2	5.4E-5	7.7E-4
Stomach wall	1.6E-2	5.7E-5	8.1E-4
Pancreas	7.6E-3	2.1E-5	3.0E-4
LLI wall	6.1E-1	2.5E-3	3.5E-2
Kidney	3.3E-1	1.0E-3	1.5E-2
Bladder wall	4.8E-3	1.2E-5	1.6E-4
ULI wall	2.0E-1	8.1E-4	1.2E-2
SI wall	3.6E-2	1.4E-4	2.0E-3
Ovaries	7.3E-3	2.1E-5	3.0E-4
Testes	8.9E-3	2.2E-5	3.1E-4
Spleen	4.6E-2	1.8E-4	2.5E-3
Uterus	7.4E-3	2.1E-5	3.0E-4
Thymus	7.9E-3	2.1E-5	3.0E-4
Thyroid	8.4E-3	2.1E-5	3.1E-4
Weighted mean	1.2E+1	6.2E-3	8.8E-2

alent rates* due to atmospheric radioactive particulate and Rn-222 emissions from the model uranium mine sites are presented in Tables 6.4 through 6.10. The Rn-222 dose equivalent rate is only for the inhalation and air immersion pathways and excludes Rn-222 daughters. The impact from Rn-222 daughters is addressed separately with a working level calculation. The dose equivalent estimates are for the model sites described for use with the AIRDOS-EPA code in Appendix K. Assumptions about food production and consumption for the maximum individual were selected for a rural setting. The maximum individual dose equivalent rate occurred about 1600 meters downwind from the center of the model site. The term "population" refers to the population living within a radius of 80 kilometers of the source. Population dose equivalents are the sum of the exposures to all individuals in the exposed population for the annual release from the model uranium mine.

Dose equivalent rates in Tables 6.4 through 6.10 indicate that the red marrow, endosteal cells, lung, kidneys, and spleen are generally the highest exposed target organs. A dose equivalent rate is presented for the "weighted mean" target organ, but this calculated result was not used in the health effect calculations. We calculated "weighted mean" dose equivalents by using organ dose equivalent weighting factors (see Appendix L) and summing the results. The weighted mean dose equivalent rate was presented instead of the total body dose equivalent rate.

Individual lifetime fatal cancer risks and estimated additional fatal cancers to the regional population due to atmospheric radioactive emissions from the model uranium mine sites are presented in Tables 6.11 and 6.12. The individual lifetime risks in Table 6.11 are those that would result from one year of exposure (external and internal) and the working levels estimated for those individuals. Except for the in situ leach mine, the individual lifetime risks in Table 6.12 are those that would result from a lifetime of exposure (71 years average life expectancy). The individual lifetime risks in Table 6.12 for the in situ leach mine are based on an exposure time of 18 years, which is the expected life, including restoration, of this type of model uranium mine.

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*The dose equivalent rates were not used to calculate risk and are only presented for perspective purposes. Risks of health impact were calculated directly from external and internal radionuclide exposure data.

Table 6.11 Individual lifetime fatal cancer risk for one year of exposure and estimated additional fatal cancers to the regional population due to annual radioactive airborne emissions from model uranium mines

Source	Maximum Exposed Individual	Average Exposed Individual	Regional Population
Average surface mine			
Particulates and Rn-222	6.7E-7	7.5E-10	1.1E-5
Radon-222 daughters	5.5E-6	1.1E-8	1.6E-4
Total	6.2E-6	1.2E-8	1.7E-4
Average large surface mine			
Particulates and Rn-222	3.7E-6	3.7E-9	5.4E-5
Radon-222 daughters	1.9E-5	4.1E-8	5.9E-4
Total	2.3E-5	4.5E-8	6.4E-4
Average underground mine			
Particulates and Rn-222	1.6E-7	2.8E-10	1.0E-5
Radon-222 daughters	1.1E-5	4.9E-8	1.7E-3
Total	1.1E-5	4.9E-8	1.7E-3
Average large underground mine			
Particulates and Rn-222	1.4E-6	2.5E-9	9.0E-5
Radon-222 daughters	1.1E-4	5.0E-7	1.8E-2
Total	1.1E-4	5.0E-7	1.8E-2
Inactive surface mine			
Particulates and Rn-222	5.5E-8	6.4E-11	9.1E-7
Radon-222 daughters	4.2E-7	8.3E-10	1.2E-5
Total	4.7E-7	8.9E-10	1.3E-5
Inactive underground mine			
Particulates and Rn-222	1.5E-8	2.0E-11	7.4E-7
Radon-222 daughters	2.7E-7	1.2E-9	4.4E-5
Total	2.8E-7	1.2E-9	4.5E-5
In situ leaching facility			
Particulates and Rn-222	1.6E-6	8.7E-10	1.2E-5
Radon-222 daughters	1.1E-5	2.1E-8	3.0E-4
Total	1.3E-5	2.2E-8	3.1E-4

Table 6.12 Individual lifetime fatal cancer risk due to lifetime exposure to radioactive airborne emissions from model uranium mines

Source	Maximum Exposed Individual	Average Exposed Individual (c)
Average surface mine(a)		
Particulates and Rn-222	1.4E-5	1.6E-8
Radon-222 daughters	1.2E-4	2.3E-7
Total	1.3E-4	2.5E-7
Average large surface mine(a)		
Particulates and Rn-222	6.6E-5	6.6E-8
Radon-222 daughters	3.5E-4	7.4E-7
Total	4.2E-4	8.1E-7
Average underground mine(a)		
Particulates and Rn-222	3.5E-6	5.8E-9
Radon-222 daughters	2.0E-4	9.0E-7
Total	2.0E-4	9.1E-7
Average large underground mine(a)		
Particulates and Rn-222	2.5E-5	4.4E-8
Radon-222 daughters	1.9E-3	8.6E-6
Total	1.9E-3	8.6E-6
Inactive surface mine(b)		
Particulates and Rn-222	3.9E-6	4.5E-9
Radon-222 daughters	3.0E-5	5.9E-8
Total	3.4E-5	6.3E-8
Inactive underground mine(b)		
Particulates and Rn-222	1.1E-6	1.4E-9
Radon-222 daughters	1.9E-5	8.5E-8
Total	2.0E-5	8.6E-8
In situ leaching facility(d)		
Particulates and Rn-222	1.6E-5	8.7E-9
Radon-222 daughters	2.0E-4	3.8E-7
Total	2.2E-4	3.9E-7

(a) Considers exposure for 17 years to active mining and 54 years to inactive mine effluents.

(b) Considers exposure for 71 years to inactive mine effluents.

(c) Considers the average individual in the regional population within an 80-km radius of the model mine.

(d) Considers 10-year operation and 8-year restoration.

Table 6.13 Genetic effect risk to descendants for one year of parental exposure to atmospheric radioactive airborne emissions from model uranium mines

Source	Descendants of Maximum Exposed Individual (effects/ birth)	Descendants of Average Exposed Individual (effects/ birth)	Descendants of Regional Population (effects/yr)
Average surface mine	6.3E-7	2.6E-9	1.6E-5
Average large surface mine	3.7E-6	1.3E-8	7.9E-5
Average underground mine	1.4E-7	2.9E-10	4.4E-6
Average large underground mine	1.1E-6	2.4E-9	3.6E-5
Inactive surface mine	6.0E-8	2.4E-10	1.4E-6
Inactive underground mine	1.6E-8	3.4E-11	5.0E-7
In situ leach facility	8.0E-9	2.7E-11	1.6E-7

Table 6.14 Genetic effect risk to descendants for a 30-year parental exposure to atmospheric radioactive airborne emissions from model uranium mines

Source	Effects/birth	
	Descendants of Maximum Exposed Individual	Descendants of Average Exposed Individual(c)
Average surface mine(a)	1.2E-5	4.6E-8
Average large surface mine(a)	6.4E-5	2.2E-7
Average underground mine(a)	2.6E-6	5.4E-9
Average large underground mine(a)	2.0E-5	4.0E-8
Inactive surface mine(b)	1.8E-6	7.2E-9
Inactive underground mine(b)	5.0E-7	5.8E-10
In situ leach facility(d)	1.4E-7	4.8E-10

(a) Considers exposure to 17 years active mining and 13 years inactive mine effluents.

(b) Considers exposure for 30 years to inactive mine effluents.

(c) Considers the average individual in the regional population within an 80-km radius of the model mine.

(d) Considers 10-year operation and 8-year restoration.

Genetic effect risks due to atmospheric radioactive emissions from the model uranium mine sites are presented in Tables 6.13 and 6.14. The risks to descendants in Table 6.13 are those that would result from one year of exposure to the parent or parents of first generation individuals. The descendant risks in Table 6.14 are those that would result from 30 years exposure to the first generation parent or parents, except for the in situ leach mine where we used an 18-year exposure time. The 30-year time period represents the mean years of life where gonadal doses are genetically significant.

We estimated the health impact risks with the DARTAB code using exposure data from the AIRDOS-EPA code. The dose equivalent and risk conversion factors that we used with the DARTAB code are tabulated in Appendix L. The somatic risk conversion factors are based on a lifetime (71 years average lifetime) exposure time, and the genetic effect risk conversion factors are based on a 30-year exposure time. When the exposure time for calculated risks was only one year, we calculated the risk by multiplying the risk calculated by DARTAB with the ratio of the one year exposure time to the exposure times used to calculate the risk conversion factors (1/71 for somatic effects and 1/30 for genetic effects to descendants of maximum and average exposed individuals).* Appendix L contains a discussion of the health risk assessment methodology.

We developed several tables to present the calculated health impact risk. The percentage contributions to the fatal cancer risks for individual sources at each model uranium mine site are contained in Table 6.15 for the maximum individual and Table 6.16 for the average individual. The fatal cancer risks by source term for one year of exposure which we used to calculate percentage contributions are contained in Tables L.4 to L.6 in Appendix L. Tables L.7 to L.9 contain genetic risks by source term at each model uranium mine site. The percent of the fatal cancer risk due to radon-222 daughter concentrations at model uranium mine sites is indicated in Table 6.17. The percent of the fatal cancer risk for principal nuclides and pathways due to radioactive particulate and Rn-222 emissions at each model uranium mine site are contained in Table 6.18.

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*A correction factor was not needed for DARTAB calculated genetic effects committed per year to the regional population.

Table 6.15 Percent of the fatal cancer risk for the maximum individual due to the sources of radioactive emissions at model uranium mines

	Percent of fatal cancer risk (a,b)				
Mine type	Mining Activities	Ore	Sub-ore	Spoils	Vehicular Dust
Average surface mine	56 (95)	18 (66)	14 (98)	12 (89)	<1 (0)
Average large surface mine	59 (93)	14 (41)	12 (98)	14 (86)	1 (0)
Average underground mine	80 (≅100)	3 (79)	17 (97)	<1 (96)	<1 (0)
Average large underground mine	89 (≅100)	2 (76)	9 (96)	<1 (96)	<1 (0)
Inactive surface mine	28(c) (100)	0	0	72 (84)	0
Inactive underground mine	77(c) (100)	0	0	23 (77)	0
In situ leach facility	100 (87)	0	0	0	0

(a) See Table L. 4, Appendix L.

(b) Values in parentheses are percent contribution of radon-222 daughters.

(c) Emissions from abandoned pit (surface mine) or vents and portals (underground mine).

Table 6.16 Percent of the fatal cancer risk for the average individual in the regional population due to the sources of radioactive emissions at model uranium mines

Mine type	Percent of fatal cancer risk(a,b)				
	Mining Activities	Ore	Sub-ore	Spoils	Vehicular Dust
Average surface mine	58 (97)	16 (78)	14 (99)	12 (93)	<1 (0)
Average large surface mine	60 (96)	11 (64)	12 (99)	16 (92)	1 (0)
Average underground mine	81 (≅100)	2 (93)	16 (99)	<1 (99)	<1 (0)
Average large underground mine	89 (≅100)	2 (91)	9 (99)	<1 (99)	<1 (0)
Inactive surface mine	29(c) (100)	0	0	71 (90)	0
Inactive underground mine	80(c) (100)	0	0	20 (92)	0
In situ leach facility	100 (96)	0	0	0	0

(a) See Table L.5, Appendix L.

(b) Values in parentheses are percent contribution of radon-222 daughters.

(c) Emissions from abandoned pit (surface mine) or vents and portals (underground mines).

Table 6.17 Percent of fatal cancer risks due to radon-222 daughter concentrations at model uranium mine sites

Source	Percent fatal cancer risk[a]
Average surface mine	89
Average large surface mine	84
Average underground mine	99
Average large underground mine	99
Inactive surface mine	88
Inactive underground mine	95
In situ leach facility	87

[a] Remainder due to radioactive particulate and Rn-222 emissions.

Table 6.18 Percent of the fatal cancer risk for principal nuclides and pathways due to radioactive particulate and Rn-222 emissions at model uranium mines

			Percent of fatal cancer risk			
			Internal Pathways		External Pathways	
Mine Type	Receptor	Principal Nuclides	Ingestion	Inhalation	Air Immersion	Ground Surface
Average Surface Mine	Max. Individual	U-238(20.0), U-234(22.1), Th-230(31.7), Ra-226(7.94), Po-210(7.33)	15.8	80.2	0.003	4.02
	Av. Individual or population	U-238(9.17), U-234(10.1), Th-230(22.7), Ra-226(21.3), Pb-210(6.92), Po-210(22.4)	60.1	38.1	0.005	1.81
Average Large Surface Mine	Max. Individual	U-238(20.0), U-234(22.2), Th-230(31.8), Ra-226(7.98)	15.9	80.0	0.002	4.05
	Av. Individual or Population	U-238(9.19), U-234(10.1), Th-230(22.7), Ra-226(21.4), Pb-210(6.94), Po-210(22.4)	60.5	37.7	0.004	1.83
Average Underground Mine	Max. Individual	U-238(17.9), U-234(19.8), Th-230(28.4), Ra-226(7.14), Po-210(6.59), Rn-222(13.6)	14.0	82.5	0.025	3.52
	Av. Individual or Population	U-238(12.0), U-234(13.2), Th-230(20.1), Ra-226(7.24), Po-210(7.31), Rn-222(34.6)	16.9	80.6	0.063	2.43
Average Large Underground Mine	Max. Individual	U-238(17.5), U-234(19.3), Th-230(27.7), Ra-226(6.97), Po-210(6.43), Rn-222(16.0)	13.6	82.9	0.029	3.39
	Av. Individual or Population	U-238(11.2), U-234(12.4), Th-230(18.8), Ra-226(6.76), Po-210(6.83), Rn-222(39.2)	15.7	82.0	0.071	2.25
Inactive Surface Mine[a]	Max. Individual	U-238(19.5), U-234(21.6), Th-230(31.0), Ra-226(9.4), Bi-214(5.31), Po-210(7.17)	16.8	75.4	0.002	7.85
	Av. Individual or Population	U-238(8.76), U-234(9.68), Th-230(21.7) Ra-226(26.1), Pb-210(6.77), Po-210(21.4)	62.2	34.3	0.003	3.48
Inactive Underground Mine[a]	Max. Individual	U-238(19.6), U-234(21.6), Th-230(31.1), Ra-226(9.45), Bi-214(5.33), Po-210(7.21)	16.9	75.2	0.001	7.88
	Av. Individual or Population	U-238(17.0), U-234(18.8), Th-230(28.5), Ra-226(12.7), Bi-214(4.81), Po-210(10.4)	26.3	66.6	0.004	7.09
In situ Leaching Facility	Max. Individual	U-238(45.2), U-234(50.0), U-235(2.21)	0.46	99.5	0.002	0.039
	Av. Individual or Population	U-238(43.3), U-234(47.8), U-235(2.12)	3.50	96.5	0.009	0.038

[a] Spoils source term only.

The fatal cancer health risk at each of the model uranium mine sites is dominated by the lung cancer risk from radon-222 daughter exposures (see Table 6.17). Radioactive particulates and Rn-222 contributed to a little over 10 percent of the total fatal cancer health risk at the model surface mines and at the in situ leaching facility (see Table 6.11). Essentially all the risks from the model underground mines are due to radon-222 daughter exposures. The fatal cancer health risks from the active model underground mines are greater than the risks from the active model surface mines because of the larger quantity of Rn-222 released. The risks are similar at inactive surface and underground mines.

The largest fatal cancer risk is from the average large underground mine (see Tables 6.11 and 6.12)--an estimated 1.9E-3 lifetime fatal cancer risk to the maximum exposed individual for a lifetime exposure. The lifetime fatal cancer risk to the average individual in the regional population is estimated to be 8.6E-6 for a lifetime exposure period. The number of estimated additional fatal cancers in the regional population per year of mine operation is estimated to be 1.8E-2.

For the active surface mines, about 60 percent of the radon daughter impact is from the exposed pit surfaces (see Table L.4). For the active underground mines, the predominate radon daughter impact is from mine vent air. For the inactive surface mine, about 70 percent of the radon daughter impact is from waste rock pile exhalation and about 30 percent was from the pit interior surfaces. About 80 percent of the radon daughter impact for the inactive underground mine was due to radon releases from the mine vents and entrance. The release of radon from the pregnant leach surge tanks was the predominate source of the radon daughter health impact risk for the model in situ leach mine. Detailed percentages of the lifetime fatal cancer risks by source term for each model uranium mine are contained in Tables 6.15 and 6.16.

The health impact from particulate radionuclides and Rn-222 was predominately due to U-238 and daughter radionuclides (see Table 6.18). Thorium-232 and daughters were only minor contributors to the particulate and Rn-222 fatal cancer risk with Rn-222 only contributing significantly (14 to 40 percent) at active underground mines. The majority of the exposure to individuals around the model uranium mines is received from the internal pathways. Inhalation was the most important internal pathway except for the average individual and regional population impact at surface mines where ingestion was the major pathway (see Table 6.18). For active surface mines, about 52 percent of the particulate and Rn-222 impact to the maximum individual was from the ore source term, and about 25 percent of the health impact was from the mining activities source term (see Table L.4). For active underground mines, between 28 and 46 percent of the particulate and Rn-222 impact was from the ore source term and between 26 and 41 percent of the particulate and Rn-222 impact was from the sub-ore source term. The predominant source of the particulate and Rn-222 impact from the inactive mines was particulate radionuclides in wind-suspended dust from the waste rock pile. The release of particulate radionuclides from the uranium recovery plant was the predominant source of the particulate health impact risk for the model in situ leach mine.

For perspective, the calculated fatal cancer risks can be compared to the estimated cancer risk from all causes. The American Cancer Society estimates the risk of cancer death from all causes to be 0.15 (Ba79). The maximum exposed individual around the model average large underground mine is estimated to incur an additional lifetime fatal cancer risk of 0.0019 (1.3 percent) due to radioactive airborne emissions from the model mine. There is a regional population of 36,004 persons for the model average large underground mine site located in New Mexico. The cancer death rate for the State of New Mexico for whites of both sexes was 154.5 deaths per year for 1973 to 1976 per 100,000 people (NCI78). Applying this statistic to the regional population, about 56 cancer deaths are estimated to occur each year in the regional population from all causes. Applying the approximate fatal cancer risk coefficient of 0.15 to the regional population of 36,004 persons, about 5,400 people in the regional area would normally die of cancer. About 0.018 additional cancer deaths (0.00033 percent) in the regional population are estimated per year of operation from radioactive airborne emissions at the model average large underground mine.

The risk of genetic effects from radiation exposure at model uranium mine sites is very small compared to the normal occurrence of hereditary disease. The national incidence of genetic effects is 60,000 per 10^6 births (NAS72). The normal occurrence of hereditary disease for the descendants of the regional population of 14,297 at the model average large surface mine in Wyoming is 0.06 effects per birth and 12.1 effects per year, based on 202 live births per year in the regional population. (We present statistics for the site of the average large surface mine since the largest genetic risk for all the evaluated model uranium mines occurred at this site [see Tables 6.13 and 6.14]). We estimated the genetic effect risk to the descendants of the maximum exposed individual to be an additional 6.4E-5 effects/birth (0.1 percent increase) for a 30-year exposure period. The genetic effect risk to the descendants of the average exposed individual in the regional population is estimated to be an additional 2.2E-7 effects/birth (0.00036 percent increase) for a 30-year exposure period. The number of additional genetic effects committed to the descendants of the regional population per year of operation of the average large surface mine is estimated to be 7.9E-5. The additional committed genetic effects constitute a very small increase to the 12.1 effects that will normally occur each year in the live births within the regional population.

6.1.2 Nonradioactive Airborne Emissions

To calculate atmospheric concentrations at the location of the maximum individual, we used the data on nonradioactive air pollutant emissions from Section 3. We compared these pollutant air concentrations with calculated nonoccupational threshold limit values, natural background concentrations, and average urban concentrations of selected airborne pollutants in the United States.

The "natural" background atmospheric concentration has been defined (Va71) as the concentration of pollutants in areas absent of activities by man which cause significant pollution. Variations in background levels may result from differences in mineral content of the soil, vegetation, wind conditions, and the proximity to the ocean or metropolitan areas. Based on an extensive literature survey and consideration of the abundance and distribution of the chemical elements in the ocean and earth's crust, a set of "natural" background airborne concentrations has been developed for the United States (Va71). Natural background airborne concentrations for selected pollutants are listed in the second column of Table 6.19. Also listed in the table are average concentrations of airborne pollutants in urban areas. The latter are arithmetic mean concentrations obtained from measurements taken over a period of several years (Va71).

6.1.2.1 Combustion Products

Airborne concentrations of combustion products released from diesel and gasoline-powered equipment were estimated for the site of the maximum

Table 6.19 Natural background concentrations and average urban concentrations of selected airborne pollutants in the United States

Pollutant	Natural Background Concentration, $\mu g/m^3$	Average Urban Concentration, $\mu g/m^3$
Gases		
CO	100	7000
NO_x	40	141
NH_3	10	80
SO_x	5	62
CO_2	594,000	NR
Hydrocarbons	NR[a]	500
Suspended particles		
Total	20 - 40	105
As	0.005	0.02 (1)
Ba	0.005	NR
Cd	0.0001	0.002
Co	0.0001	0.0005
Cr	0.001	0.015
Cu	0.01	0.09
Hg	0.0005	0.1
Fe	0.2 - 0.5	1.58
Pb	0.001	0.79
Mg	0.1	NR
Mn	0.01	0.1
Mo	0.0005	0.005
Ni	0.001	0.034
Se	0.001	NR
Sr	0.005	NR
Th	0.0005	NR
U	0.0001	NR
V	0.001	0.05
Zn	0.01	0.67
Zr	0.001	NR

[a] NR - Not Reported.

Source: Va71; except for CO_2, Ba76.

individual. The concentrations were computed using the annual release rates given in Tables 3.30 and 3.52 with dispersion parameters applicable for the model underground (New Mexico) and surface (Wyoming) mining areas (Appendix K). The estimated combustion product concentrations are low compared to the natural background and average urban concentrations (see Table 6.20). A conservative threshold limit value (TLV) was computed, as described in Section 6.1.2.3 for SO_2, CO, and NO_2. Of these pollutants, only the nitrogen oxide concentrations at the average large surface mine exceed the nonoccupational TLV. Considering these comparisons and the conservative nature of the analyses, combustion products released from heavy uranium mining equipment do not appear to pose a health hazard.

6.1.2.2 Nonradioactive Gases

Airborne concentrations of the three principal nonradioactive gases released from the hypothetical in situ leach mining site were computed using the source terms from Table 3.59 and the meteorological parameters and dispersion model described in Appendix K. Table 6.21 shows the estimated atmospheric concentrations at the location of a maximum individual; occupational threshold limit values (TLV's); adjusted TLV's applicable to nonoccupational exposures; and the percent the estimated concentrations are of the adjusted TLV's. The occupational TLV's have been conservatively adjusted. They were adjusted on the basis of a 168-hr week, instead of a 40-hour week and a safety factor of 100.

The results of this analysis indicate that two of the estimated concentrations fall below their respective TLV's, and the concentration of ammonium chloride is approximately equal to its TLV. Considering the conservative nature of the adjusted nonoccupational TLV on which the comparisons were made, none of the nonradioactive gases appear to be at concentrations that might pose a serious health hazard. The ammonia level is about 80 percent of the estimated "natural" background concentration and only about 10 percent of the average urban concentration (Table 6.19).

6.1.2.3 Trace Metals and Particulates in the Form of Dust

We identified seventeen trace metals and particulates in the form of dust as potential airborne emissions from uranium mines. Table 6.22 presents projected airborne concentrations of the metals and particulates at the site of the maximum individual for six mine classifications. As might

Table 6.20 Combustion product concentrations at the site of the maximum individual with comparisons, $\mu g/m^3$

Pollutant	Average underground mine	Average large underground mine	Average surface mine	Average large surface mine	Natural background concentration[(a)]	Average urban concentration[(a)]	Non-occupational TLV[(b)]
Particulates of combustion	1.4E-3	1.6E-2	9.7E-2	4.5E-1	NR[(c)]	NR	NR
SO_x	1.2E-2	1.3E-1	5.5E-1	2.2E+0	5E+0	6.2E+1	3.1E+1
CO	9.7E-2	1.1E+0	4.3E+0	1.8E+1	1.0E+2	7.0E+3	1.3E+2
NO_x	1.6E-1	1.8E+0	7.1E+0	3.0E+1	4.0E+1	1.4E+2	2.1E+1
Hydrocarbons	1.6E-2	1.8E-1	7.1E-1	3.1E+0	NR	5.0E+2	NR

(a) See Table 6.19.

(b) Nonoccupational TLV = TLV (mg/m^3) x 40 hr/168 hr x 10^{-2} x 10^3 $\mu g/mg$ (ACGIH76).

(c) NR - Not reported.

Table 6.21 A comparison of the airborne concentrations of nonradioactive gases at the hypothetical in situ leach site with threshold limit values

Contaminant	Atmospheric Concentration(a) (μ g/m^3)	TLV(b) (mg/m^3)	Non-occupational(c) TLV (μg/m^3)	Percent of Nonoccupational TLV
NH_3	8.1	18	43	19
NH_4Cl	24	10	24	100
CO_2	60	9000	21,400	0.3

(a)Location of maximum individual.

(b)Source: ACGIH76.

(c)Nonoccupational TLV = TLV (mg/m^3) x 40 hr/168 hr x 10^{-2} x 10^3 μg/mg.

be expected, large surface mine emissions usually have the greatest concentrations, and those from inactive underground mines the least. Projected metal concentrations range from a low of about 5×10^{-7} μgm/m^3 of cobalt from inactive underground mines to a high of about 1 μgm/m^3 of potassium from large surface mines.

Table 6.23 shows where particulates (dust) or trace metal air concentrations are estimated to exceed natural background or average urban air concentrations (Table 6.19). Several trace metal air concentrations exceed "natural" background; however, only the estimated air concentration of particulates (dust) exceeds the air concentration of airborne pollutants in urban areas.

We evaluated the significance of these concentrations by comparing them with threshold limit values (TLV's) for workroom environments published by the American Conference of Governmental Industrial Hygienists (ACGIH76). These TLV's, which are for occupational workers and a 40-hour workweek, were adjusted by multiplying by 40/168 to convert them to continuous exposure values and dividing by 100 to make them applicable to the general public. Table 6.24 is a tabulation of the adjusted TLV's, the projected concentrations of metals and particulates (from Table 6.22), and the ratio of these concentrations to the adjusted TLV's. The sums of these ratios provide a measure of whether a mixture of the metals would be a significant problem, a sum greater than one indicating that the "composite" TLV has been exceeded.

Table 6.24 shows that in no case does a single metal exceed its TLV, nor do any of the mixtures exceed a "composite" TLV. Although TLV's were not available for potassium and strontium, their low toxicity and low concentrations make it unlikely that their addition to the sums would change this conclusion. For the worst case, large surface mines, the sum of ratios is only about 17 percent of the limit.

Particulates, on the other hand, present a different picture. The TLV for nonspecific particulates, nuisance dust, was chosen for comparison. It can be seen that the TLV is exceeded by a factor of six at the large model surface mine and nearly exceeded at the average model surface mine. About 50% of the exposure to dust is from vehicular traffic, and about 30% results from mining activities within the pit.

In summary, specific trace metal airborne emissions from uranium mines do not appear to present a significant hazard, either singly or as com-

Table 6.22 Stable trace metal airborne concentrations at the site of the maximum individual, $\mu g/m^3$

Trace metal	Avg. underground mine	Avg. large underground mine	Avg. surface mine	Avg. large surface mine	Inactive underground mine	Inactive surface mine
As	3.1E-5	1.9E-4	2.6E-4	1.5E-3	3.1E-6	1.5E-5
Ba	5.1E-4	1.8E-3	7.0E-3	4.2E-2	3.6E-5	1.6E-4
Co	4.0E-6	3.1E-5	1.1E-5	4.7E-5	4.5E-7	2.9E-6
Cu	3.3E-5	1.5E-4	4.4E-4	2.6E-3	2.2E-6	1.1E-5
Cr	5.0E-5	1.4E-4	1.1E-3	6.9E-3	8.9E-7	3.6E-6
Fe	9.7E-3	4.1E-2	1.4E-1	8.5E-1	6.3E-4	2.7E-3
Hg	7.2E-6	1.5E-5	1.8E-4	1.1E-3	NA[a]	NA
K	1.3E-2	6.3E-2	1.7E-1	1.0	9.8E-4	4.4E-3
Mg	9.4E-4	6.9E-3	2.5E-3	1.0E-2	1.3E-4	6.2E-4
Mn	7.1E-4	2.8E-3	1.1E-2	6.8E-2	3.7E-5	1.7E-4
Mo	3.3E-5	2.3E-4	1.4E-4	6.7E-4	4.5E-6	2.0E-5
Ni	4.9E-6	3.9E-5	1.4E-5	5.8E-5	8.9E-7	3.6E-6
Pb	4.1E-5	2.0E-4	5.4E-4	3.2E-3	3.1E-6	1.4E-5
Se	3.1E-5	2.2E-4	1.2E-4	5.9E-4	4.5E-6	1.9E-5
Sr	1.7E-4	5.5E-4	3.4E-3	2.1E-2	4.9E-6	2.3E-5
V	4.7E-4	3.0E-3	2.2E-3	1.8E-2	5.4E-5	2.5E-4
Zn	2.6E-5	9.6E-5	4.6E-4	2.8E-3	1.3E-6	5.2E-6
Part[b]	1.2	3.9	2.3E+1	1.4E+2	3.9E-2	1.7E-1

[a] NA - Not available.

[b] Part. - Particulates (dust).

Table 6.23 Comparison of stable trace metal airborne concentrations at the location of the maximum individual with natural background concentrations and average urban concentrations of these airborne pollutants

Exceed Natural Background[a]	Exceed Average Urban Concentration[a]
Average Large Surface Mine	
Ba, Cr (possible), Fe, Hg (possible), Mn, Mo, Pb, Sr, V, particulates	Particulates
Average Surface Mine	
Ba, Cr (possible), Mn, V	None
Average Large Underground Mine	
V	None
Average Underground Mine	
None	None

(a) See Tables 6.19 and 6.22.

Table 6.24 Comparison of trace metal airborne concentrations at the site of the maximum individual with threshold limit values (TLV's) in the workroom environment adjusted for continuous exposure to the general public, $\mu g/m^3$

Trace metal	Adjusted[(a)] TLV	Average Underground Mine		Average Large Underground mine		Average Surface Mine		Average Large Surface mine		Inactive Underground mine		Inactive Surface mine	
		Conc.	Conc./TLV	Conc.	Conc./TLV	Conc.	Conc./TLV	Conc.	Conc./TLV	Conc.	Conc./TLV	Conc.	Conc./TLV
As	1.2	3.1E-5	3E-5	1.9E-4	2E-4	2.6E-4	2E-4	1.5E-3	1E-3	3.1E-6	3E-6	1.5E-5	1E-5
Ba	1.2	5.1E-4	4E-4	1.8E-3	2E-3	7.0E-3	6E-3	4.2E-2	4E-2	3.6E-5	3E-5	1.6E-4	1E-4
Co	0.24	4.0E-6	2E-5	3.1E-5	1E-4	1.1E-5	5E-5	4.7E-5	2E-4	4.5E-7	2E-6	2.9E-6	1E-5
Cu	0.48	3.3E-5	7E-5	1.5E-4	3E-4	4.4E-4	9E-4	2.6E-3	5E-3	2.2E-6	5E-6	1.1E-5	2E-5
Cr	1.2	5E-5	4E-5	1.4E-4	1E-4	1.1E-3	9E-4	6.9E-3	6E-3	8.9E-7	7E-7	3.6E-6	3E-6
Fe	12	9.7E-3	8E-4	4.1E-2	3E-3	1.4E-1	1E-2	8.5E-1	7E-2	6.3E-4	5E-5	2.7E-3	2E-4
Hg	0.12	7.2E-6	6E-5	1.5E-5	1E-4	1.8E-4	2E-3	1.1E-3	9E-3	NA	----	NA	--
K	NA[(b)]	1.3E-2	---	6.3E-2	---	1.7E-1	--	1.0E+0	--	9.8E-4	----	4.4E-3	--
Mg	24	9.4E-4	4E-5	6.9E-3	3E-4	2.5E-3	1E-4	1.0E-2	4E-4	1.3E-4	5E-6	6.2E-4	3E-5
Mn	12	7.1E-4	6E-5	2.8E-3	2E-4	1.1E-2	9E-4	6.8E-2	6E-3	3.7E-5	3E-6	1.7E-4	1E-5
Mo	12	3.3E-5	3E-6	2.3E-4	2E-5	1.4E-4	1E-5	6.7E-4	6E-5	4.5E-6	4E-7	2.0E-5	2E-6
Ni	0.24	4.9E-6	2E-5	3.9E-5	2E-4	1.4E-5	6E-5	5.8E-5	2E-4	8.9E-7	4E-6	3.6E-6	2E-5
Pb	0.36	4.1E-5	1E-4	2.0E-4	6E-4	5.4E-4	2E-3	3.2E-3	9E-3	3.1E-6	9E-6	1.4E-5	4E-5
Se	0.48	3.1E-5	6E-5	2.2E-4	5E-4	1.2E-4	2E-4	5.9E-4	1E-3	4.5E-6	9E-6	1.9E-5	4E-5
Sr	NA	1.7E-4	--	5.5E-4	--	3.4E-3	--	2.1E-2	--	4.9E-6	--	2.3E-5	--
V	1.2	4.7E-4	4E-4	3.0E-3	2E-3	2.2E-3	2E-3	1.8E-2	2E-2	5.4E-5	4E-5	2.5E-4	2E-4
Zn	12	2.6E-5	2E-6	9.6E-5	8E-6	4.6E-4	4E-5	2.8E-3	2E-4	1.3E-6	1E-7	5.2E-6	4E-7
Total of ratios			2E-3		1E-2		3E-2		1.7E-1		2E-4		7E-4
Particulates:													
Dust	24[(c)]	1.2E+0	5E-2	3.9E+0	2E-1	2.3E+1	1E+0	1.4E+2	6E+0	3.9E-2	2E-3	1.7E-1	7E-3

(a) Adjusted TLV = Occupational TLV (mg/m^3) x 40 hr/168hr x 10^3 µg/mg x 1/100.

(b) NA - Not available.

(c) Limit for nuisance dust - total mass.

Source: Workroom TLV's from ACGIH76.

posite mixtures, when evaluated against adjusted threshold limit values. However, particulate emissions, at least for surface mines, require further evaluation. If model predictions can be verified by measurement, control measures are indicated.

6.1.3 Radioactive Aquatic Emissions

We used the data on radioactive releases from mine dewatering (Sections 3.3.3 and 3.4.3) to estimate the public health impact of mining operations at a typical active underground mining site (New Mexico) and a typical active surface mining site (Wyoming). The health risks estimated in this section are of fatal cancers and genetic effects to succeeding generations. Dose equivalents and health risks per year of active mine operation are estimated for the maximum and average individuals and for the population of each assessment area. These calculated dose equivalents and health risk estimates are believed to be higher than the actual dose equivalents and health risks because of the conservative assumptions required to predict movement of radionuclides in surface waters (see Section J.2 of Appendix J). Very few data are available on aquatic releases from inactive mines; hence, the significance of these releases, particularly for Colorado and Utah where inactive mines are numerous, could not be determined.

The individual and population dose equivalents presented in this section are computed using the models and parameters discussed in Appendix J. The health risk estimates are generated by the following procedures:

a. For inhalation or ingestion of radionuclides, the quantity of radionuclides taken into the body is determined as part of the dose equivalent calculations. This quantity is multiplied by a health risk per unit intake conversion factor.

b. For external irradiation from ground deposited radionuclides or from air submersion, the dose equivalents are calculated and multiplied by a health risk per unit dose equivalent conversion factor.

The health risk per unit intake and health risk per unit external dose equivalent conversion factors for aquatic releases are listed in Tables J.13 and J.14, Appendix J. This appendix also discusses the health risk assessment methodology used to obtain the risks presented in this section. Uranium and Ra-226 releases are given for both active mining sites. It is assumed that the stated uranium releases are entirely U-238 and that U-234 is in equilibrium with the U-238 but that Th-230 precipitates out of the mine water.

Table 6.25 Annual radiation dose equivalent rates due to aquatic releases from the New Mexico model underground mine

Organ	Maximum Individual Dose Rate (mrem/y)	Average Individual Dose Rate (mrem/y)	Population Dose Rate (person-rem/y)
Endosteal	5.6E+1	5.0	3.2E+2
Red Marrow	2.0	1.6E-1	1.1E+1
Lung	1.3	2.1E-3	1.4E-1
Liver	5.5E-1	2.9E-2	1.9
Stomach Wall	1.9E-1	3.8E-3	2.5E-1
LLI Wall(a)	9.4E-1	6.6E-2	4.3
Thyroid	4.5E-1	2.5E-2	1.6
Kidney	2.8E+1	2.4	1.6E+2
Muscle	4.9E-1	2.5E-2	1.6
Ovaries	4.1E-1	2.4E-2	7.8E-1
Testes	4.7E-1	2.4E-2	7.8E-1
Weighted Mean	2.2	1.5E-1	9.9

(a)Lower large intestine wall.

Table 6.26 Annual radiation dose equivalent rates due to aquatic releases from the Wyoming model surface mine

Organ	Maximum Individual Dose Rate (mrem/y)	Average Individual Dose Rate (mrem/y)	Population Dose Rate (person-rem/y)
Endosteal	6.8E-1	2.1E-1	3.4
Red Marrow	3.8E-2	7.4E-3	1.2E-1
Lung	2.3E-2	1.0E-4	1.7E-3
Liver	3.0E-2	2.8E-3	4.5E-2
Stomach Wall	1.0E-2	2.8E-4	4.6E-3
LLI Wall(a)	2.9E-2	7.7E-3	1.3E-1
Thyroid	1.8E-2	1.4E-3	2.3E-2
Kidney	4.0E-1	1.1E-1	1.8
Muscle	1.9E-2	1.5E-3	2.4E-2
Ovaries	1.5E-2	1.5E-3	1.2E-2
Testes	1.8E-2	1.4E-3	1.2E-2
Weighted Mean	4.0E-2	7.1E-3	1.2E-1

(a) Lower large intestine wall.

Table 6.27 Individual lifetime fatal cancer risk and committed fatal cancers to the population residing within the assessment areas

Source	Maximum exposed individual lifetime fatal cancer risk for operation of the mine		Average exposed individual lifetime fatal cancer risk for operation of the mine[(a)]		Committed fatal cancers for the assessment area population for operation of the mine	
	1 yr.	17 yrs.	1 yr.	17 yrs.	1 yr.	17 yrs.
Underground mine site (New Mexico)	3.3E-7	5.6E-6	2.0E-8	3.4E-7	1.3E-3	2.2E-2
Surface mine Site (Wyoming)	7.1E-9	1.2E-7	9.6E-10	1.6E-8	1.6E-5	2.7E-4

(a) The average individual risk is the cumulative population risk divided by the population residing within the assessment area.

Also, it is assumed that Rn-222, Pb-214, Bi-214, Pb-210, and Po-210 are in equilibrium with the Ra-226. For example, a reported release rate of 0.01 Ci/yr of U-238 would be reflected in the analyses as 0.01 Ci/yr of U-238 and 0.01 Ci/yr of U-234. In like manner, a release of 0.001 Ci/yr of Ra-226 would be reflected in the analyses as 0.001 Ci/yr Ra-226, 0.001 Ci/yr Rn-222, 0.001 Ci/yr Pb-214, 0.001 Ci/yr Bi-214, 0.001 Ci/yr Pb-210, and 0.001 Ci/yr Po-210.

The maximum individual, average individual, and population annual dose equivalent rates due to release of mine water containing radionuclides are given in Tables 6.25 and 6.26 for the two active uranium mine sites. The population dose equivalent rates are the sum of the dose equivalent rates to all individuals residing within the assessment areas due to the annual release from the model uranium mine. Average individual dose equivalent rates are computed by dividing the population dose equivalent rates by the number of persons in the assessment area.

The dose equivalent rates in Tables 6.25 and 6.26 indicate that the endosteal cells and kidney are the highest exposed target organs. Ingestion is the predominant exposure mode for both the endosteal cells and the kidney.

Individual lifetime fatal cancer risks and committed fatal cancers to the population within the assessment area for radionuclide releases due to mine dewatering are presented in Table 6.27. The maximum and average individual lifetime risks (columns 2 and 3, respectively) and the committed fatal cancers to the population within the assessment area (column 4) are shown for both one year of release of radionuclides due to mine dewatering and, in parenthesis, for the cumulative release over the 17 years of mine operation. To compute the 17-year risks, the one-year risks are multiplied by 17, which assumes equal annual radionuclide discharges. At both the model underground (New Mexico) and surface (Wyoming) mines, the majority of the risk is from releases of U-238, U-234, and Po-210.

A perspective on the additional fatal cancers estimated for the population (Table 6.27) can be gained by realizing that the probability of an individual dying of cancer of all types is 0.15 (Ba79). Taking the New Mexico assessment area (64,950 persons) as an example, the expected number of deaths from all forms of cancer for this population is 9,743 persons. For the 17 years of mine operations, the estimated increase in the number of deaths from cancer in the assessment area population is 0.022 deaths

(Table 6.27). This represents a 0.00023 percent increase in the expected fatal cancer occurrences in the assessment area population as a result of operation of the underground mine in New Mexico over its 17-year active life. For the Wyoming assessment area (16,230 persons), the estimated increase in the expected fatal cancer deaths due to operation of the surface mine for 17 years is 0.000011 percent.

Table 6.28 presents the genetic risks to succeeding generations, for exposure to both individuals and the population within the assessment area, caused by mine dewatering radionuclide releases. The genetic risks to succeeding generations of maximum and average exposed individuals (columns 2 and 3, respectively) and the committed genetic effects to the descendants of the present population within the assessment area (column 4) are shown for one year of releases. The mechanics and assumptions used to estimate the genetic effects are similar to those used to estimate fatal cancer risks (see Appendix J). For both the model underground (New Mexico) and surface (Wyoming) mines the majority of the risk is from releases of U-238, U-234, and Po-210.

The risks of additional genetic effects due to the discharge of contaminated mine water from model uranium mine sites are very small when compared to the normal occurrence of hereditary diseases. As given in Section 6.1.1, the natural incidence of genetic effects is 60,000 per million births (NAS72), or 0.06 effects per birth. This natural incidence rate is equivalent to 848 effects per year per million persons, considering a birth rate of 0.01413 births per person-year. Taking the New Mexico site as an example, the normal incidence of genetic effects for the assessment area population (64,950 persons) during the 17 years of operation of the mine would be 936 genetic effects. The increase in genetic effects committed to the assessment area population during the 17 years of operation is 0.015 genetic effects committed. Thus, the genetic effects committed due to aquatic wastes released during the operation of the New Mexico underground mine are only 0.0016% of the genetic effects which occur due to other causes during the mine operating life. For the Wyoming site (16,230 persons), the genetic effects committed due to aquatic wastes released during the operation of the model surface mine are only 0.0001% of the genetic effects which occur due to other causes during the mine operating life. It

Table 6.28 Genetic risks to succeeding generations of an individual and committed genetic effects to descendants of the present population residing within the assessment area

Source	Genetic effects committed to succeeding generations of an individual for operation of the mine for 1 year[a]		Genetic effects committed to the descendants of the present population for operation of the mine for 1 year
	Maximum Individual	Average Individual	
Underground mine site (New Mexico)	4.5E-7	3.3E-8	9.0E-4
Surface mine site (Wyoming)	1.4E-8	2.0E-9	1.4E-5

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[a] Genetic effects assume 1 birth per person.

should be noted that genetic effect risks to descendants of individuals cannot be added to somatic effect risks for these individuals.

6.1.4 Nonradioactive Aquatic Emissions

Data on nonradiological emissions from uranium mines via the water pathway are limited. Table 6.29 presents available estimates of concentrations of four trace metals plus sulfate and suspended solids in discharge streams from the model surface mine located in Wyoming and seven trace metals plus sulfate and suspended solids from the model underground mine located in New Mexico. These concentrations are calculated after dilution in the first order tributaries (Appendix J) and represent average concentrations for the assessment areas. The concentrations presented in Table 6.29 are conservative since, with the exception of sulfates, loss of contaminants due to precipitation, adsorption, and infiltration to shallow aquifers are not considered. The concentrations are calculated by diluting discharges from a mine into the first order surface streams with no losses. For sulfate, a more realistic approach is taken since only 20 percent of it is assumed to remain in solution in the surface stream, as discussed in Section 3.3.3.1.4.

Also presented in Table 6.29 are recommended agricultural water concentration limits for livestock and irrigation for several of these elements (EPA73). Drinking water limits are not presented because public water supplies are normally derived from groundwater rather than surface water, so drinking water would not be a pathway of concern for the average individual in the assessment area. Though drinking water would be a potentially significant pathway for the maximum individual, the data available for this analysis did not allow a reliable prediction of groundwater concentrations due to mine dewatering (Appendix J). For this reason, the impact of nonradioactive waterborne emission on the maximum exposed individual could not be evaluated. The ratios of the average water concentrations to these limits are also listed in Table 6.29 and show that only molybdenum from the underground mine approaches its limit (irrigation). Also, the sums of the ratios being less than one indicate that mixtures of the metals would not exceed a "composite limit" for an average individual in the assessment area.

Table 6.29 Comparison of nonradiological waterborne emissions from uranium mines with recommended agricultural water quality limits

Parameter	Recommended Limits, mg/ℓ		Model Surface Mine			Model Underground Mine		
	Livestock	Irrigation	Avg. Water Conc., mg/ℓ	Ratio Avg./Livestock Limit	Ratio, Avg./ Irrigation Limit	Avg. Water Conc., mg/ℓ	Ratio Avg./Livestock Limit	Ratio, Avg./ Irrigation Limit
Arsenic	0.2	0.1	1.4E-4	0.0007	0.0014	3.1E-4	0.0016	0.0031
Barium	NA[a]	NA				2.0E-2		
Cadmium	0.05	0.01	1.1E-4	0.0022	0.011	1.6E-4	0.0032	0.016
Molybdenum	NA	0.01				7.0E-3		0.70
Selenium	0.05	0.02				1.6E-3	0.032	0.08
Zinc	25	2.0	4.8E-4	0.00002	0.00024	1.1E-3	0.00004	0.00055
Uranium	NA	NA	2.0E-3			3.5E-2		
Sulfate	NA	NA	4.9			2.9		
Total suspended solids	NA	NA	5.8E-1			6.8E-1		
Totals				0.0029	0.013		0.037	0.80 (0.1)[b]

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[a] NA - Not available.

[b] Excluding molybdenum.

Because of the limited number of data available, it is difficult to evaluate the significance of these discharges. Although molybdenum could be a problem, it is not possible to quantify the risk from molybdenum to the maximum individual without having estimates of drinking water concentrations. Uranium, the metal estimated to be in highest concentration (Table 6.29), has no established limits based on chemical toxicity in the United States. In Canada, the maximum acceptable concentration for uranium in drinking water based on chemical toxicity has been set at 0.02 mg/ℓ (0.04 mg/day), considering a continuous lifetime intake rate of 2 liters of water per day (HWC78). It is reasonable to assume that limits for uranium in water used for irrigation and to water livestock would exceed the drinking water limit. Hence, based on the estimated uranium concentrations at surface (0.002 mg/ℓ) and underground (0.035 mg/ℓ) uranium mines, the water would probably be acceptable for irrigation and livestock watering. The other constituents, such as solids and sulfates, for which limits are not available, have minimal or no toxic properties.

It is premature to conclude the health hazard caused by non-radiological waterborne emissions from uranium mines. Before definitive conclusions can be reached, additional information is needed. Of particular interest would be data on water use patterns in the vicinity of the mines and the degree to which the mine discharges may infiltrate groundwater supplies.

6.1.5 Solid Wastes

6.1.5.1 Radium-226 Content

Solid wastes, consisting of sub-ore, waste rock, and overburden, at active and inactive uranium mines contain elevated concentrations of radium-226.* The sub-ore may contain as much as 100 pCi/g of radium-226. Even though the overburden and waste rock contain lower concentrations than the sub-ore, most of these wastes contain concentrations of radium-226 in quantities greater than 5 pCi/g (see Sections 3.3.1, 3.4.1, 3.7.1, and 3.7.2).

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* The radium-226 concentration in natural soil and rock is about 1 pCi/g.

Uranium mine wastes containing radium-226 in quantities greater than 5 pCi/g have been designated as "hazardous wastes" in a recently proposed EPA regulation (43FR58946, December 18, 1978) under the Resource Conservation and Recovery Act (RCRA). This is primarily due to the fact that the use of these wastes under or around habitable structures could significantly increase the chance of lung cancer to individuals occupying these structures.

6.1.5.2 Estimates of Potential Risk

We have estimated the risk of fatal lung cancer that could occur to individuals living in houses built on land contaminated by uranium mine wastes (Table 6.30). Risks were estimated for homes built on land containing radium-226 soil concentrations ranging from 5 to 30 pCi/g. The relationship between the indoor radon-222 decay product concentration and the radium-226 concentration in soil under a structure is extremely variable and depends upon many complex factors. Therefore, the data in Table 6.30 only illustrate the levels of risks that could occur to individuals living in structures built on contaminated land. These data should not be interpreted as establishing a firm relationship between radium-226 concentrations in soil and indoor radon-222 decay product concentrations.

Table 6.30 Estimated lifetime risk of fatal lung cancer to individuals living in homes built on land contaminated by uranium mine wastes

^{226}Ra in Soil (pCi/g)	Indoor Working Levels (WL)	Lifetime Risk of Fatal Lung Cancer[a] (per 100 persons)
5	0.02	2.5
10	0.04	5.0
20	0.08	10
30	0.12	15

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[a] Based on an individual being inside the home 75 percent of the time.

The working level concentrations in Table 6.30 were derived from calculations made by Healey (He78), who estimated that 1 pCi/g of radium-226 in underlying loam-type soil would result in about 0.004 WL inside a house with an air change rate of 0.5 per hour. These calculated working levels are in reasonable agreement with measurements made by EPA (Fig. 6.1) at 21 house sites in Florida (S.T. Windham, U.S. Environmental Protection Agency, Written Communication, 1980). The Florida data were derived from the average radium-226 concentration in soil (core samples were taken to a maximum depth of three feet at each site) and the average radon-222 decay product concentration inside each structure.

6.1.5.3 Using Radium Bearing Wastes In The Construction of Habitable Structures

Wastes containing elevated levels of radium-226 have been used at a number of locations in the construction of habitable structures. In Grand Junction, Colorado, uranium mill tailings were widely used as landfill under and around the foundations of homes and other structures causing high radon-222 decay product concentrations inside many structures. To remedy this situation, Public Law 92-314 was passed in 1972 to establish a federal-state remedial action program to correct the affected structures. In Mesa County, Colorado, which includes Grand Junction, uranium mill tailings were identified at about 6,000 locations. About 800 of these locations are expected to receive corrective action because the radon decay product concentrations inside buildings constructed at these locations exceeded the remedial action criteria (DOE79). According to the criteria, dwellings and schoolhouses would be recommended for remedial action if the indoor radon decay product concentration exceeded 0.01 WL above background; other structures would be recommended for remedial action if the indoor radon decay product concentration exceeded 0.03 WL above background.

In central Florida, structures have been built on reclaimed phosphate land. The reclaimed land is composed of phosphate mining wastes that contain elevated radium-226 concentrations. EPA estimates that about 1,500 to 4,000 residential or commercial structures are located on 7,500 acres of the total 50,000 acres of reclaimed phosphate-mined lands (EPA79). A survey of 93 structures built on reclaimed phosphate land showed that about 40 percent of the structures had indoor radon-222 decay product concentrations in excess of 0.01 WL and about 20 percent had concentrations in

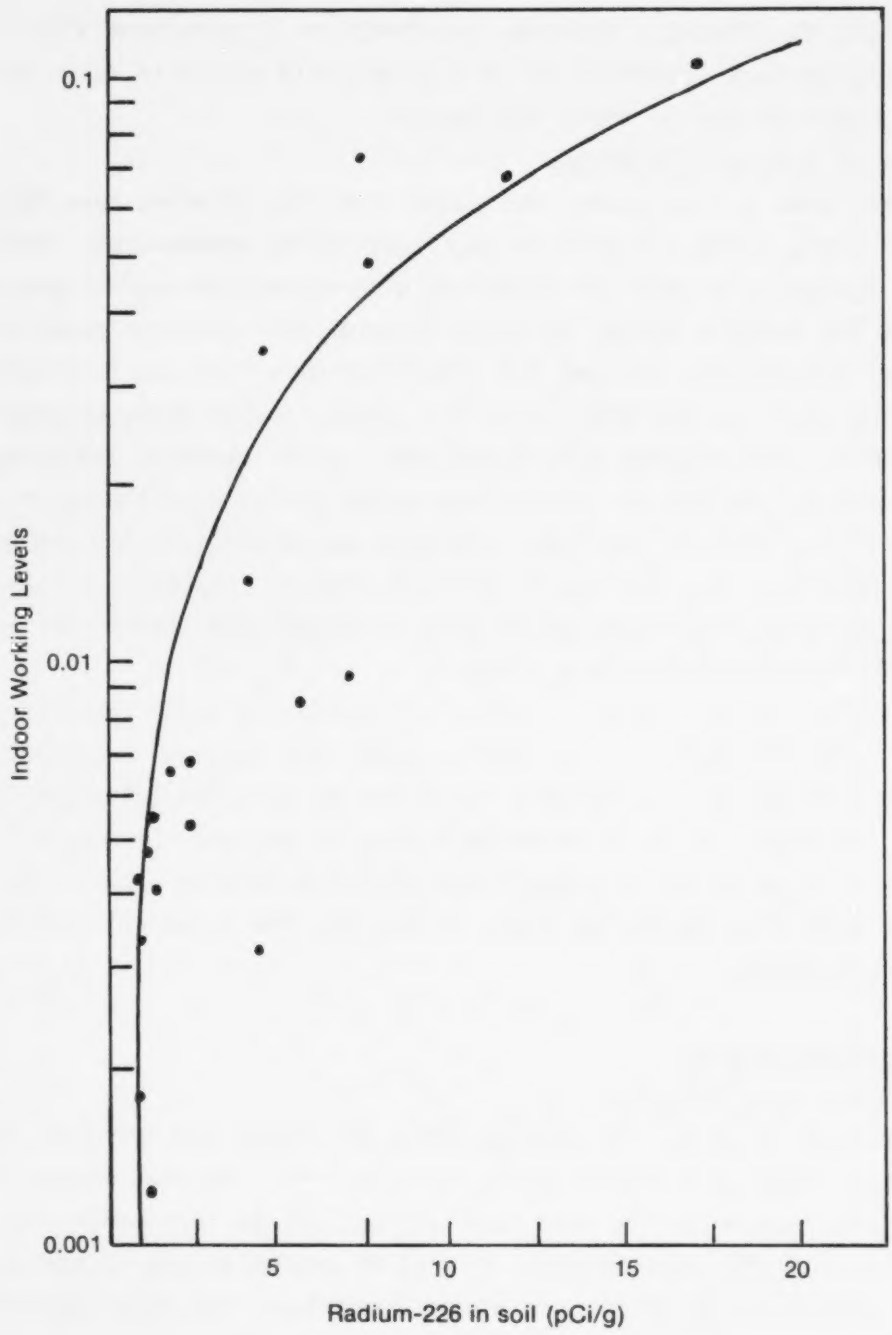

Figure 6.1 Average indoor radon-222 decay product measurements (in working levels) as a function of average radium-226 concentration in soil.

excess of 0.03 WL (EPA79). Lifetime residency in a structure with a radon-222 decay product concentration of 0.03 WL could result in twice the normal 3 to 4 percent risk of fatal lung cancer.

6.1.5.3.1 Use of Uranium Mine Wastes

We do not know to what extent the wastes from uranium mines have been removed from mining sites and used in local and nearby communities. However, while surveying in 1972 for locations with higher-than-normal gamma radiation in the Western States to locate uranium mill tailings material used in local communities, EPA and AEC identified more than 500 locations where "uranium ore" was believed to be the source of the elevated gamma radiation (ORP73). The specific type of ore (mill-grade, sub-ore, low-grade waste rock) was not determined as this was beyond the scope of the survey. At some locations, however, surveyors attempted to characterize the ore by using such terms as "ore spillage," "ore specimens," "low-grade crushed ore," or "mine waste dump material." Some locations were identified as sites of former ore-buying stations (ORP73).

Since it is unlikely that valuable mill-grade ore would have been widely available for off-site use, we suspect that uranium mine waste (perhaps sub-ore) may be the source of the elevated gamma radiation levels at many of the locations where large quantities of ore material are present. Table 6.31 shows the locations where higher-than-normal gamma radiation levels were detected during these surveys and the suspected sources of the elevated levels.

6.2 Environmental Effects

6.2.1 General Considerations

Minerals are necessary to augment man's existence and welfare; in order to obtain them, some form of mining is necessary. The very nature of mining requires disturbing the land surface, but may be considered transitory. To discuss the environmental effects of uranium mining in particular, it is convenient to divide the mining operations into three phases. The first phase includes the exploration for, and the delineation of, the ore body. This involves, in most cases, substantial exploratory and development drilling. The second phase involves the preparation of the mine site and the mining process itself. This phase includes the construction of service areas, dewatering impoundments, and access roads, digging or drilling of mine entries, etc. During the actual mining process, waste

Table 6.31 Gamma radiation anomalies and causes

Location	Number of Anomalies Detected	Cause of Anomaly				
		Tailings	Uranium Ore	Radioactive Source	Natural Radioactivity	Unknown
Arizona						
Cane Valley[a]	19	15	4			
Cameron	3		1			2
Cutter	5		4	1		
Tuba City	17	7			3	7
State Total	44	22	9	1	3	9
Colorado[b]						
Cameo	3	1				2
Canon City	187	36	24		99	28
Clifton	1083	159	31	3	14	876
Collbran	145	4	2			139
Craig	86	8	7		46	25
Debeque	109	2			1	106
Delta	43	1	3		29	10
Dove Creek	83	59	17	2	2	3
Durango	354	118	18	49	67	102
Fruita	1276	58	47	1	26	1144
Gateway	17	12	1	1		3
Glade Park	1	1				
Grand Valley	110	10	2			98
Gunnison	47	3	8	1	28	7
Leadville	91	18	2		65	6
Loma	199	10	3	1	4	181
Mack	90	6	2	1		82
Mesa	123	1	1			120
Mesa Lakes	3					3
Molina	43					43
Naturita	33	10	15	5	1	2
Nucla	13	3	6		2	2
Palisade	939	107	36	3	14	779
Plateau City	28	1				27
Rifle	810	168	20	7	1	614
Salida	64	6	2		52	4
Slick Rock	9	3	5	1		
Uravan	209	208				1
Whitewater	55		4		2	49
State Total	6253	1013	256	75	453	4456
Idaho						
Idaho City	3				2	1
Lowman	12	9			3	
Salmon	77	1	2		65	9
State Total	92	10	2		70	10

Table 6.31 (continued)

Location	Number of Anomalies Detected	Cause of Anomaly				
		Tailings	Uranium Ore	Radioactive Source	Natural Radioactivity	Unknown
New Mexico						
Bluewater	2	1	1			
Gamerco	5				5	
Grants	101	7	49	1	25	19
Milan	41	5	23	4	1	8
Shiprock	9	8	1	0		
State Total	158	21	74	5	31	27
Oregon						
Lakeview	18		2		10	6
New Pine Creek	4		1			3
State Total	22		3		10	9
South Dakota						
Edgemont	55	43	2	1	1	8
Hot Springs	45		3		17	25
Provo	4	3		1		
State Total	104	46	5	2	18	33
Texas						
Campbellton	7		1		6	
Coughran	1				1	
Falls City	5	2			3	
Fashing	1		1			
Floresville	16				14	2
George West	10				10	
Karnes City	10	2			6	2
Kenedy	22	1	1		13	7
Panna Maria	3				3	
Pawnee	1		1			
Pleasanton	21		1	2	17	1
Poth	15				14	1
Three Rivers	5	1			2	2
Tilden	11				11	
Whitsett	1				1	
State Total	129	6	5	2	101	15

Table 6.31 (continued)

	Number of Anomalies	Cause of Anomaly				
Location	Detected	Tailings	Uranium Ore	Radioactive Source	Natural Radioactivity	Unknown
Utah						
Blanding	38	10	21		3	4
Bluff	2		1			1
Cisco	2		2			
Crescent Junction	2		1			1
Green River	23	1	14		1	7
Magna	27	1	1	1	21	3
Mexican Hat	5		4	1		
Mexican Hat (Old Mill)	14	10	1	2	1	
Moab	125	15	76	7	6	21
Monticello	59	31	16	3		9
Salt Lake City(c)	225	70	10	5	76	64
Thompson	30	26	3	0		1
State Total	552	164	150	19	108	111
Washington						
Creston	3				3	
Ford	1				1	
Reardan	10				10	
Springdale	2				2	
State Total	16				16	
Wyoming						
Hudson	8		2		5	1
Jeffery City	28	13	9	1	3	2
Lander	86	4	8	1	53	20
Riverton	86	15	14	1	33	23
Shirley Basin	9	9				
State Total	217	41	33	3	94	46
Totals	7587	1323	537	107	904	4716

(a) From EPA report ORP/LV-75-2, August 1975. Cane Valley was not included in initial gamma survey program.

(b) Excluding Grand Junction where non-tailings anomalies were not sub-categorized according to source.

(c) Salt Lake City was not completely surveyed.

Source: ORP73.

piles are produced, mine vents drilled or reamed, and pits opened and sometimes closed. In the third or retirement phase, the site is subject to deterioration from weathering ad infinitum. The extent of the deterioration depends somewhat on the amount and quality of reclamation conducted during this phase.

6.2.2 Effects of Mine Dewatering

Both surface and underground mines are dewatered in order to excavate or sink shafts and to penetrate and remove the ore body. Dewatering is by ditches, sumps, and drill holes within the mine or by high capacity wells peripheral to the mine and associated shafts. Dewatering rates up to 4 x 10^5 m^3/day have been reported in the literature. Average discharge for the surface and underground mines modeled herein are 3.0 and 2.0 m^3/minute/mine, respectively. Between 33 and 72 new mines are projected in the San Juan Basin of New Mexico alone. Total annual discharge is expected to exceed 1.48 x 10^9 m^3. Calculated effects include decreased flow in the San Juan (0.05 m^3/min) and the Rio Grande (0.85 m^3/min) rivers. Future mining will be primarily underground and the average mine depth will increase 275 percent, i.e., from 248 m to 681 m. Average mine discharge is expected to increase from 2.42 m^3/min to 13.8 m^3/min.

Aside from the hydraulic and water quality effects of discharging copious quantities of mine water to typically ephemeral streams, dewatering impacts are receiving increasing scrutiny because of the observed and calculated impacts on regional water availability and quality. Declines of water levels in regionally-significant aquifers of New Mexico and reduced base flow to surface streams are expected. Water quality effects relating to inter-aquifer connection and water transfer as a result of both dewatering and exploratory drilling have not been evaluated in any uranium mining area. In several Texas uranium districts, the effects of massive dewatering associated with surface mining are beginning to receive attention, but definitive studies have not yet begun and regulatory action is not expected in the near future. With respect to in situ leach mining, dewatering is not necessary and hence is not a concern. There is, however, some question concerning the practice of pumping large volumes of groundwater to restore aquifers. It is likely that both dewatering and aquifer restoration practices will come under increasing State regulation in water-short areas, particularly in areas of designated groundwater basins or where aquifers connect with fully-appropriated surface streams. The uncertainties surrounding environmental impacts of mining in this area can be expected to increase, and additional, comprehensive investigations of the effects of mine dewatering and wastewater discharge are needed. Expansion in Wyoming and Texas surface and in situ leaching operations is similar, and these areas should be included in future investigations.

Uranium in water removed from mines through deliberate pumping or gravity flow is extracted for sale when the concentration is 2 to 3 mg/ℓ or more. If there is subsequent discharge to surface water, radium-226 is also removed down to concentrations of 2 to 4 pCi/ℓ to comply with NPDES permit conditions. Use of settling ponds at the mines also reduces total suspended solids and may reduce other dissolved constituents as a result of aeration and coprecipitation. Seepage from such settling ponds is believed to be low and, therefore, environmentally insignificant relative to groundwater. Management of waterborne solid wastes is inconsistent from one mine to another. In some cases, the solids are collected and put in with mill tailings, but in most cases they remain at the mine portal and are covered over.

For surface versus underground mines, we recognize certain inconsistencies in the parameters chosen to calculate contaminant loading of streams. Contaminant loadings from a model surface uranium mine were calculated for uranium, radium, TSS, sulfate, zinc, cadmium, and arsenic. As noted in Section 3.3.1, molybdenum, selenium, manganese, vanadium, copper, zinc, and lead are commonly associated with uranium deposits; however, there were too few data for the latter elements to develop an "average" condition. In addition, barium, iron, and magnesium can be abundant in New Mexico uranium deposits. There were insufficient data for these elements in the case of surface uranium mines in Wyoming, hence contaminant loadings were not calculated. Regional differences dictate which parameters are monitored for baseline definition and NPDES purposes. Not all potential contaminants are important in every region. For this reason and others, State and industry monitoring programs are inconsistent with respect to parameters. Since the scope of this study did not permit extensive field surveys, maximum reliance was placed on published, readily-available data.

In terms of parameters and concentrations, NPDES permit limits are inconsistent from one EPA Region to another and from one facility to another in a given Region. In part, this reflects previous screening of the effluent discharge data and natural variations in the chemistry of ore bodies.

However, the inconsistencies in parameters included and concentration limits are sufficiently large as to suggest reevaluating NPDES permits and specifying more consistent limits that more closely reflect contaminant concentrations and volumes of mine discharge.

Infiltration of most of the mine discharge in Wyoming and New Mexico is confirmed by field observations from these States. The modeling results agree with these field data. Furthermore, the modeling results, i.e., maximum infiltration, are consistent with those in the generic assessment of uranium milling (NRC79). Potable aquifers are defined under the Safe Drinking Water Act as those which contain less than 10,000 mg/ℓ TDS. Shallow groundwater throughout the uranium regions of the U.S. meets this criterion.

Considering that essentially all of the mine effluent infiltrates and is a source of recharge to shallow potable aquifers, NPDES limits should be influenced by the drinking water regulations and ambient groundwater quality. The latter is essentially never considered with respect to mine discharges. Extensive use of soils in both the saturated and unsaturated zones as sinks for significant masses of both water and toxic chemical constituents originating in the mine discharge necessitates further evaluation of the fate of these elements. Present understanding of fractionation and resuspension processes affecting stable and radioactive trace elements greatly limits accurate prediction of health and environmental effects of mine discharge.

6.2.3 Erosion of Mined Lands and Associated Wastes

Increased erosion and sediment yield result from mining activities ranging from initial exploration through the postoperative phase. Access roads and drilling pads and bare piles of overburden/waste rock and sub-ore constitute the most significant waste sources. Dispersal is by overland flow originating as precipitation and snowmelt. To a lesser extent, wind also transports wastes and sub-ore to the offsite environment. Underground mining is much less disruptive to the surface terrain than is surface mining. Documentation of the processes and removal rates is scarce and consists of isolated studies in Texas, Wyoming, and New Mexico. Conservatively assuming that sediment yields characteristic of the areas containing the mines also apply to the mine wastes, yields of overburden, waste rock, ore, and sub-ore amount to 90,000 m^3 per year. Total sediment yield from all mining sources, including exploration and development activities, is estimated at 6.3 x 10^6 m^3.

Actual erosion rates from specific sources could be considerably above or below this value owing to such variables as pile shape and slope, degree of induration and grain size, vegetative cover, and local climatic patterns and cycles. Slope instability does present serious uranium mine waste problems throughout the mountainous uranium mining areas of Colorado (S.M. Kelsey, State of Colorado, written communication, 1979). Field observations in four western states confirm that some erosion characterizes essentially every pile but that proper reclamation, particularly grading and plant cover, provides marked improvement and may actually reduce sediment loss to below pre-mining levels. Unstabilized overburden, waste rock, and sub-ore piles revegetate rather slowly, even in areas of ample rainfall such as south Texas.

Stable trace metals such as molybdenum, selenium, arsenic, manganese, vanadium, copper, zinc, and lead are commonly associated with uranium ore and may cause deleterious environmental and health effects. Mercury and cadmium are rarely present. There is no apparent relationship between the concentration of trace metals and ore grade. In New Mexico ores, selenium, barium, iron, potassium, magnesium, manganese, and vanadium are most abundant. Presently, very few data are available to characterize the trace metal concentrations in overburden rock. Results of trace metal analyses of a few grab samples from several uranium mines in New Mexico and one in Wyoming show that except for selenium, vanadium, and arsenic, no significant trend attributable to uranium mining was present (N.A. Wogman, Battelle Pacific Northwest Laboratory, Written Communication, 1979). Considering the background concentration for these elements and the limited number of analyses, the inference of offsite contamination based on these elements is indefinite.

Ore storage piles, used to hold ore at the mine for periods averaging one month, are potential sources of contamination to the environment via dusts suspended and transported by the wind, precipitation runoff, and Rn-222 exhalation--all of which can be significantly reduced by proper management. Similarly, spoil piles remaining as a result of overburden, waste rock, or sub-ore accumulations left on the land surface after mining constitute a source of contaminants for transport by wind and water. Waste particles enriched in stable and radioactive solids and Rn-222 can be transported by wind and precipitation runoff. Such transport can be reduced through proper grading and installation of soil covers protected by vegetation or rip-rap.

Soil samples collected from ephemeral drainage courses downgrade from inactive uranium mines in New Mexico and Wyoming generally revealed no significant offsite movement of contaminants (See Appendix G). For the New Mexico mines studied, Ra-226 was elevated to about twice local background at distances of 100 to 500 meters from the mine. Water and soil samples from a surface mining site in Wyoming showed no significant offsite movement of mine-related pollution although some local transport of stockpiled ore was evident in drainage areas on and immediately adjacent to one mine pit. The strongest evidence that mine wastes are a source of local soil and water contamination is the radiochemical data and uranium in particular. Substantial disequilibrium between radium and uranium may indicate leaching and remobilization of uranium, although disequilibrium in the ore body is also suspect.

6.2.4 Land Disturbance from Exploratory and Development Drilling

About 1.3 x 10^6 exploratory and development drill holes have been drilled through 1977 by the uranium mining industry (see Section 3.6.1). Using the estimated land area of 0.51 hectares disturbed per drill hole (Pe79), about 6.5 x 10^5 hectares of land have been disturbed by drilling through 1977. To further refine the estimates of land areas disturbed, we reviewed some recent drilling areas at three mine sites. From observing 187 recent drill sites, it was concluded that 0.015 ± 0.006 hectares per drill pad were physically disturbed. The error term for the estimates is at the 95 percent confidence level. The land area disturbed by roads to gain access to the drill sites was also estimated from aerial photography and amounts to 0.17 ± 0.11 hectares. The error term for this estimate is also at the 95 percent confidence level. The total area disturbed per drill site (drill pad and access roads) is 0.19 ± 0.11 hectares. Using the latter estimates from aerial photography, the total land area disturbed from all drilling through 1977 ranges from about 1000 to 4000 km^2 with a mean of about 2500 km^2. Drilling wastes removed from the boreholes can disturb additional land areas through wind and water erosion. Ore and sub-ore remaining in the drilling wastes can, in a radiological sense, disturb land areas around the drill site from erosion. The extent of the

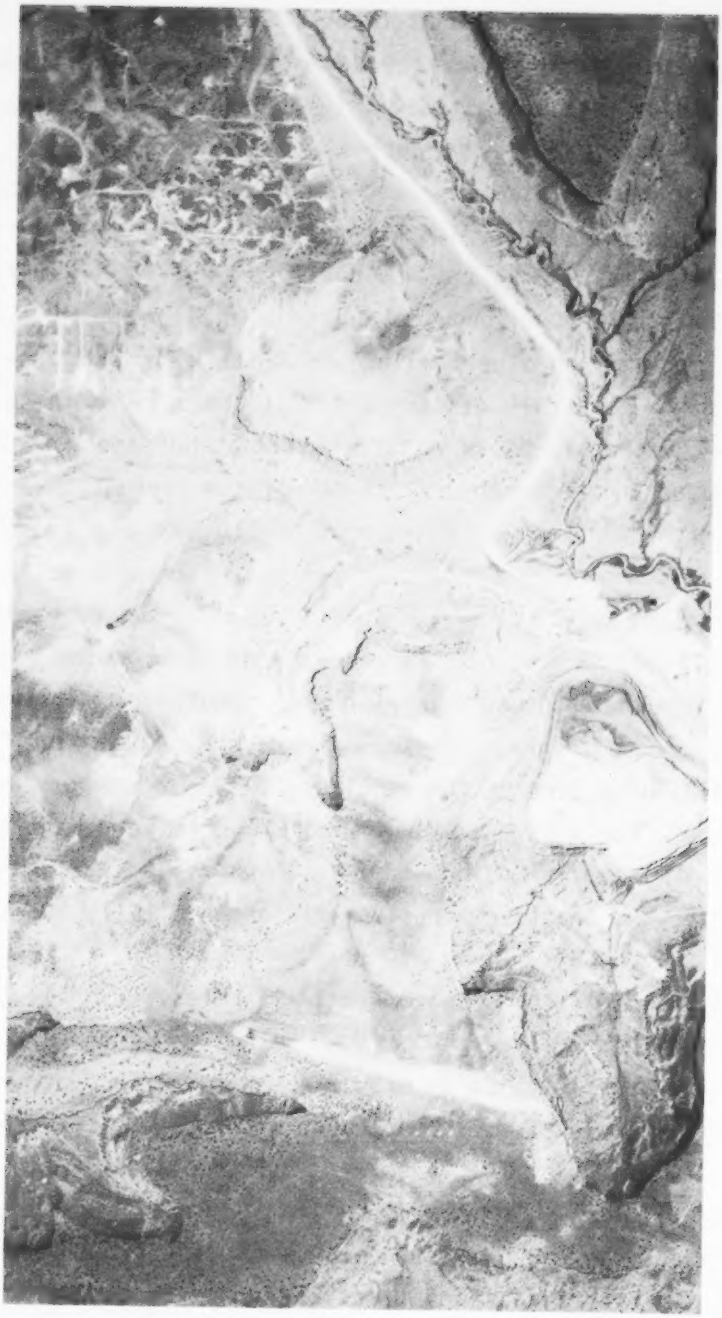

Figure 6.2 Example of natural reclamation of drill sites.

radiological contamination at drill sites is not known and cannot presently be estimated.

Some reversal of the initial environmental damage at older drill sites was also observed from aerial photographs. Figure 6.2 contains a typical medium-to-large surface uranium mine and some adjacent drilling areas that show the effects of weathering. New drill sites are in the upper left-hand corner of the photograph. The access roads and drill pads are plainly visible. It also appears that exposed drilling wastes remain at the drill site. The area left of center in the photograph shows drill sites that are probably intermediate in age. The drilling wastes remaining have very little voluntary vegetation growing on them, and appear to have been subject to wind erosion. Weathering of the drill pads and access roads is obvious, as they are hardly discernible. It appears, in these cases, that weathering may be considered a natural reclamation phenomenon. Old drill holes are located in the lower left corner of the photograph. The drilling wastes appear to be isolated dots; the drill pads and roads are almost indistinguishable from the surrounding terrain. It appears that weathering and volunteer plant growth tend to obscure scarring caused by roads located in relatively level areas. In Figure 6.3, an underground mine site, the access roads to the adjacent drill sites required extensive excavation because of the topography. These more severe excavation "scars" will probably remain for a long period of time.

In summary, the average number of drill holes per mine can be estimated by dividing the total number of holes drilled through 1977 by the number of active and inactive mines in existence in 1977:

$$\frac{1.3 \times 10^6 \text{ drill holes}}{3300 \text{ mines}} \approx 400 \frac{\text{drill holes}}{\text{mine}}. \qquad (6.1)$$

The total land area physically disturbed from drilling per mine is

$$400 \frac{\text{drill holes}}{\text{mine}} \times \frac{0.19 \text{ hectares}}{\text{drill hole}} \times \frac{\text{km}^2}{100 \text{ hectares}} = \frac{0.76 \text{ km}^2}{\text{mine}}. \qquad (6.2)$$

In some instances, weathering and volunteer plant growth (natural reclamation) tend to restore the land areas disturbed by drilling. In others, especially on rugged topography where extensive excavation has occurred, weathering may promote extensive erosion rather than natural reclamation. Any ore or sub-ore remaining at the drill sites is subject to erosion.

6.2.5 Land Disturbance from Mining

6.2.5.1 Underground Mines

At underground mines, some land area must be disturbed to accommodate equipment, buildings, wastes, vehicle parking, and so on. The disturbed area may range widely between mines in the same area or in different geographical areas. The land area disturbed by 10 mines was estimated from aerial photographs. Nine of the mines were in New Mexico and one was in Wyoming. The disturbed land area averaged 9.3 hectares per mine site and ranged from 0.89 to 17 hectares. Access roads for each mine site consumed about 1.1 hectares on the average and ranged from 0.20 to 2.59 hectares. Subsidence or the collapse of the underground workings also causes some land disturbances. An estimated 2.8 km^2 of land has subsided as a result of uranium mining in New Mexico from 1930-71 (Pa74). A crude estimate of the land disturbed from subsidence per mine can be made by dividing the subsided area by the number of inactive underground mines in New Mexico. This amounts to about 1.5 hectares per mine. The total area (mine site, access roads, and subsidence) disturbed by an underground mine is estimated to be 12 hectares.

6.2.5.2 Surface Mines

An estimate of land disturbed from surface mining was also made from aerial photographs of eight mining sites in New Mexico and two in Wyoming. The area estimates are for a single pit or a group of interconnected pits, including the area covered by mine wastes. The average disturbed area was estimated to be about 40.5 hectares and ranged from 1.1 to 154 hectares. Access roads for the pits averaged 2.95 hectares (0.03 km^2) and ranged from 0.18 to 18 hectares. The total area disturbed per mine site is about 44 hectares.

6.2.6 Retirement Phase

The actual exploration and mining of the uranium ore constitutes a very small portion of the total existence time of a mine when considered over a large time frame. The natural forces of erosion and weathering, as well as plant growth, will eventually change any work or alterations that man has made on the landscape. For example, underground mines may eventually collapse and fill with water if they are in a water table; waste piles erode and disperse in the environment; the sharp edges of pits become

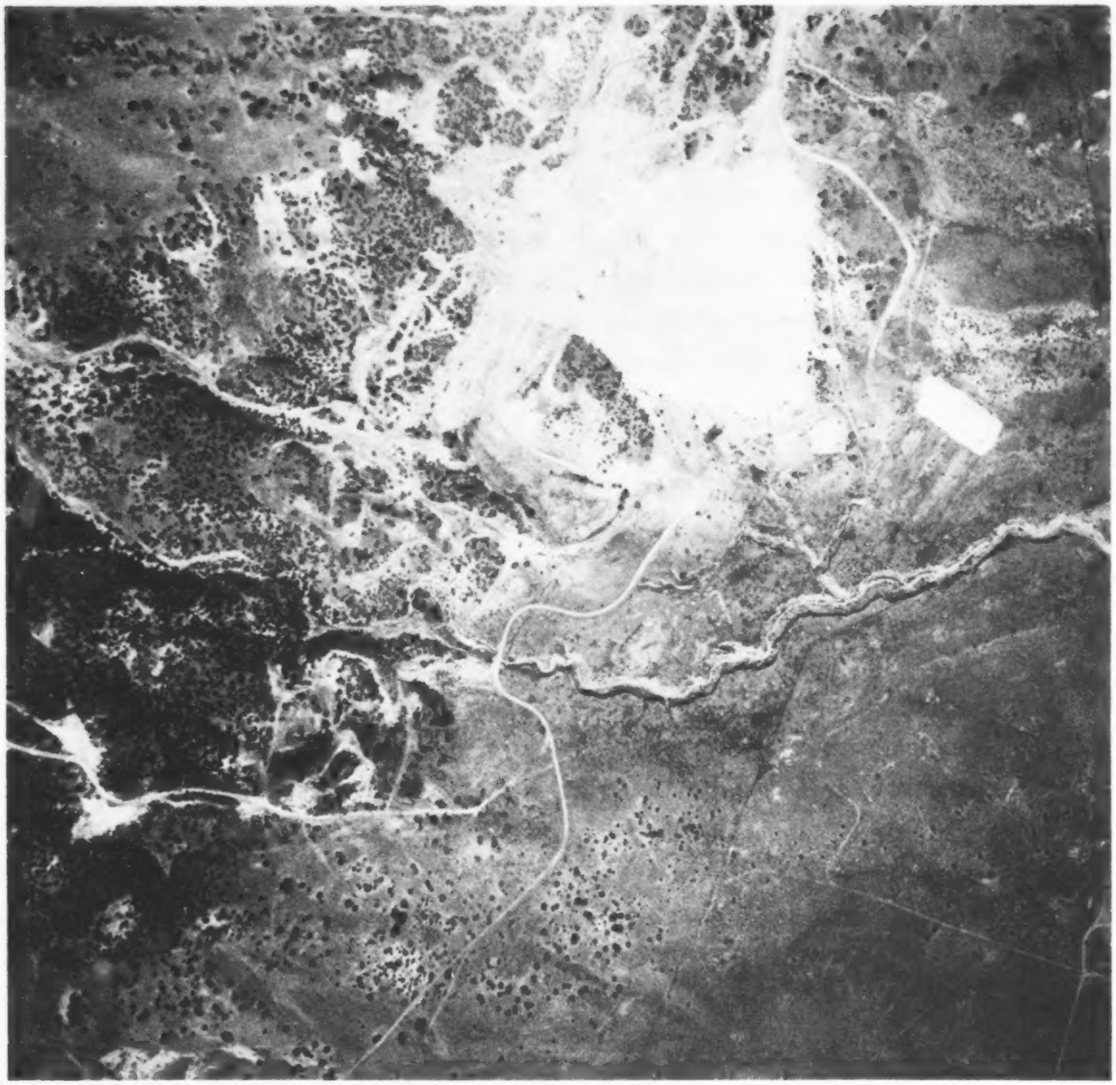

Figure 6.3 Inactive underground mine site.

smooth from wind and water erosion; lakes that are produced in pits fill up with sediment; vents and mine entries collapse, etc.

Perhaps one of the more important considerations associated with allowing a mine site to be naturally reclaimed is the dispersal of the mine wastes. Their removal from underground and subsequent storage on the surface constitute a technological enhancement of both radioactive materials and trace metals, creating a low-level radioactive materials disposal site. It appears that containment of the wastes would be preferred over their dispersal. Wastes from underground mines deposited near the entries are subject to substantial erosion. Figure 6.3 is an aerial photograph of an inactive underground uranium mine. The large light area is the waste pile and the small pile nearby is a heap-leach area. Erosion is occurring on both. A possible solution to this problem is to minimize the amount of wastes brought to the surface by backfilling mined-out areas. Another technique to minimize the dispersal of wastes into the environment by containment is to stabilize them. Unfortunately, a substantial quantity of wastes from past mining activities have been dumped in depressions and washes, which, in essence, enhances their dispersion into the environment. In retrospect, the wastes should have been stored in areas where minimal erosion would occur and then covered with sufficient topsoil to promote plant growth.

In surface mining, radiological containment can be accommodated by keeping the topsoil, waste rock, and sub-ores segregated during their removal. When backfilling, the materials can be returned to the pit in the order they were removed or in an order that would enhance the radiological quality of the ground surface. In this manner, the wastes would be contained and essentially removed from the biosphere. Figure 6.4 shows some examples of inactive and active surface mines. Some weathering and natural revegetation are noticeable around the inactive pits. Revegetation, on the other hand, appears to be relatively sparse at other inactive pits.

Erosion in inactive mining areas in New Mexico and Texas can result in deep gullying of mine waste and overburden piles. The mine wastes blanketing the foreground of Figure 6.5 are incised by an ephemeral stream that has been subsequently crossed by a roadbed in the immediate foreground. This particular mine, located in the Mesa Montanosa area immediately south of Ambrosia Lake, New Mexico, was active from 1957 to 1964. Thus, erosion occurred in about 15 years. In the background is a large mine waste pile, the toe of which is being undercut by the same ephemeral stream (Fig. 6.6). No deliberate revegetation of the mine wastes dumped in either discrete piles or spread over the landscape (Fig. 6.7) is occurring, due in large part to the unfavorable physical and chemical characteristics of the wastes. The wastes are devoid of organic matter and are enriched in stable and radioactive trace elements, some of which are toxic to plant life. Low rainfall and poor moisture retention characteristics further suppress vegetative growth. As shown in Fig. 6.7, there is a sharp contrast between the vegetative cover on mine wastes versus that on the undisturbed rangeland in the background. Waste rock from many if not most of the mines in New Mexico, Utah, and Colorado is weakly cemented sandstone with numerous shale partings. Physical breakdown to loose, easily-eroded soil unsuitable for plant life is common (Fig. 6.8), and transport by overland flow and ephemeral streams occurs both during and long after the period of active mining (Fig. 6.9).

Depending on the degree of reclamation, if any, inactive surface mines in Texas vary considerably in the degree of erosion and revegetation. For example, the deep gullying shown in Fig. 6.10 developed in a period of one year. The mine wastes in this case were not contoured or covered to minimize gamma radiation, excessive erosion, or revegetation. In fact, the wastes were disturbed and shifted very recently in the course of constructing the holding pond (for mine water pumped from an active mine to the right of the picture) in the background. Drainage in this instance is internal, i.e., to a holding pond. In the background are more recent mine waste piles also showing deep gullying, scant vegetation, and lack of protective soil covering. Mine wastes in Texas are not completely returned to the mine primarily because of the excessive cost. As in the case of most mining operations, the bulking factor makes it physically impossible to completely dispose of the wastes in the mines.

Surface mines in Texas, particularly the older ones, also have associated overland flow to the offsite environment. Shown in Fig. 6.11 is a principal channel floored by unstabilized mine wastes and draining toward nearby grazing lands. Numerous deer and doves also were observed in the area and are actively pursued by sportsmen. The unstabilized mine in this photograph was last active several years ago, but most activity stopped in 1964. Vegetation has been very slow to reestablish and is essentially limited to a very hardy, drought-resistant willow shown in the center of the picture.

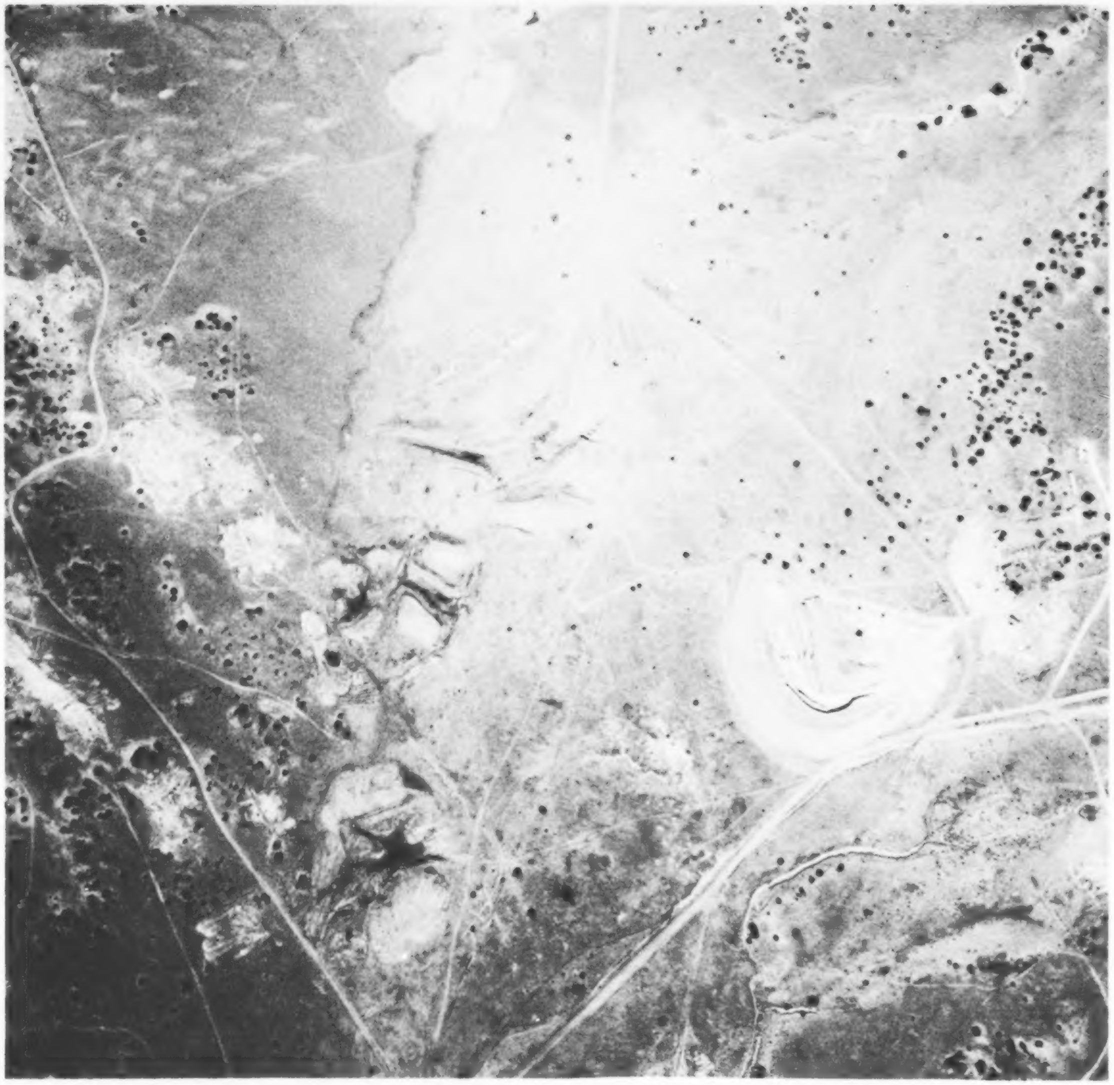

Figure 6.4 Example of active and inactive surface mining activities.

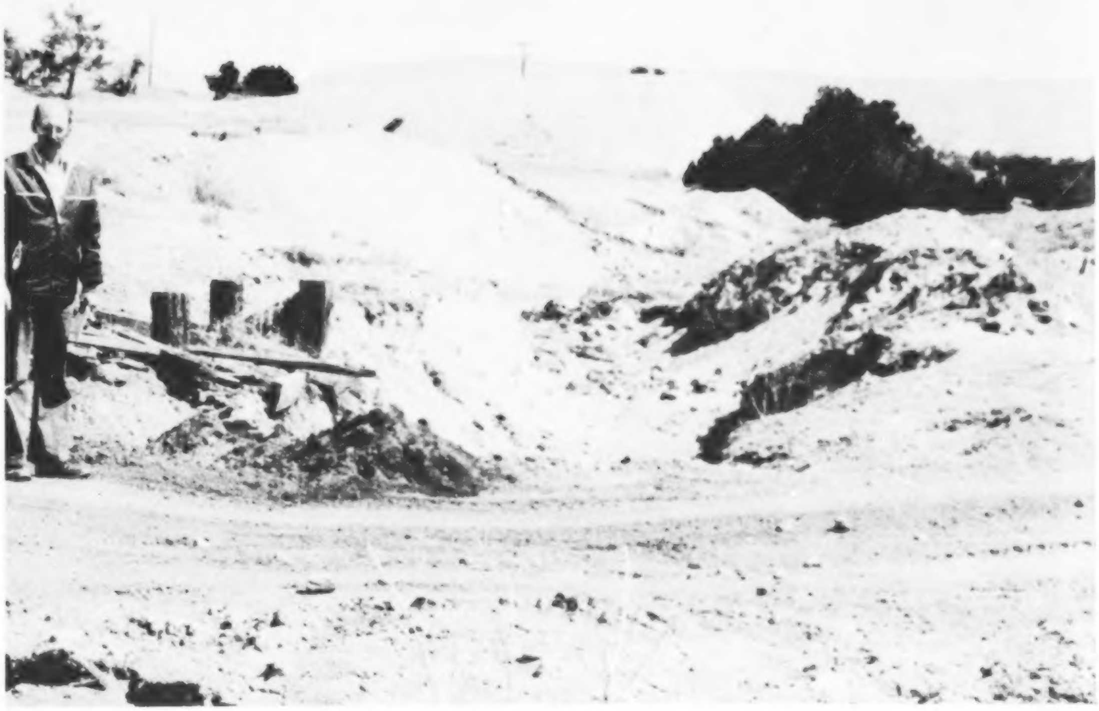

Figure 6.5 Mine wastes eroded by ephemeral streams in the Mesa Montanosa area, New Mexico.

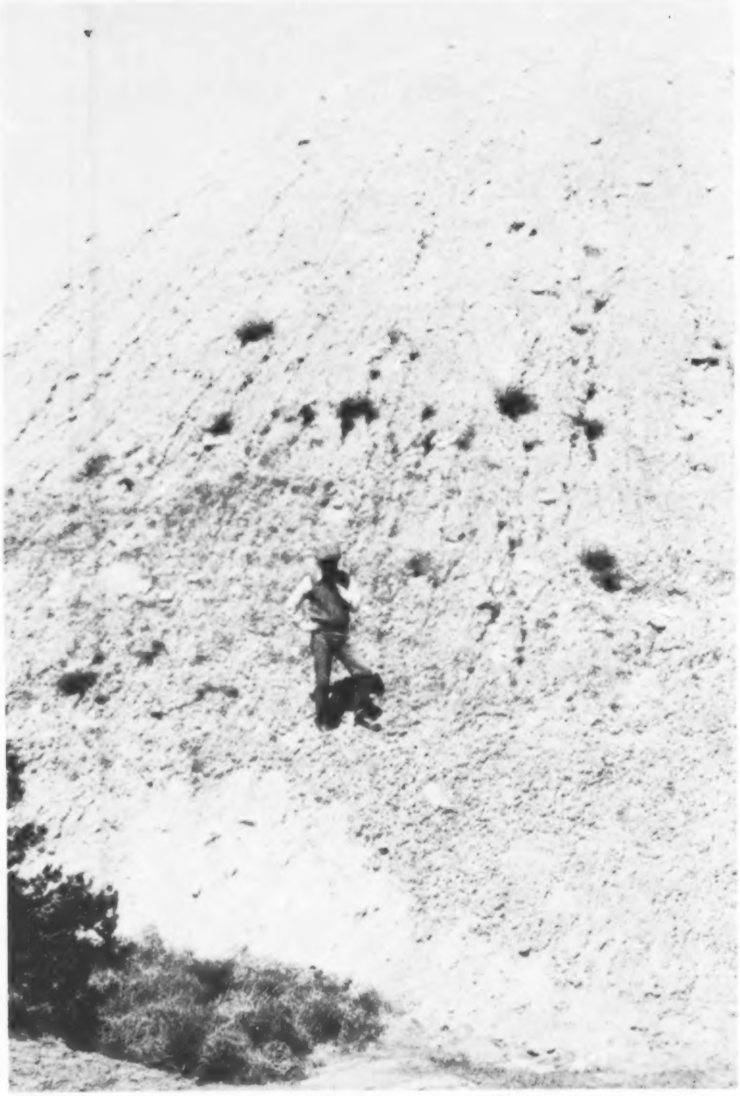

Figure 6.6 Basal erosion of a uranium mine waste pile by an ephemeral stream in the Mesa Montanosa area, New Mexico.

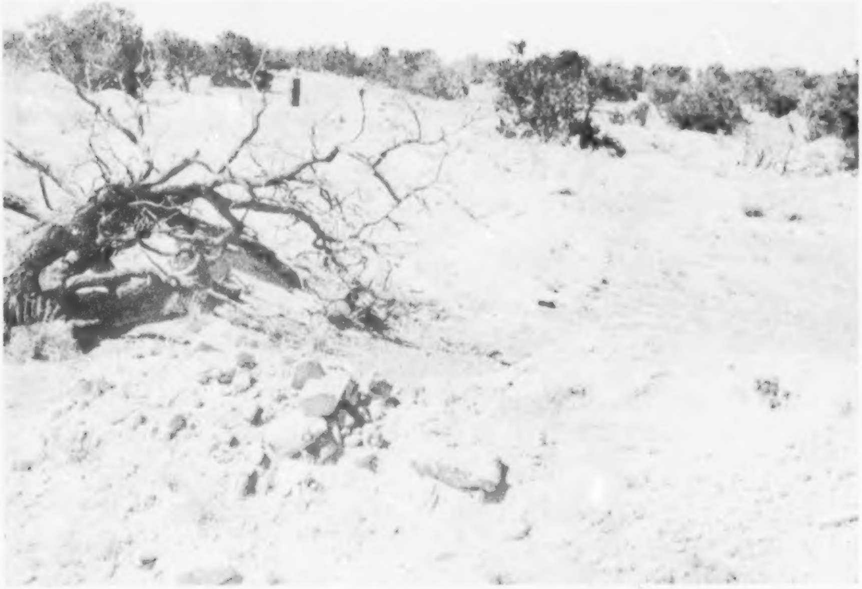

Figure 6.7 Scattered piles of mine waste at the Mesa Top Mine, Mesa Montanosa, New Mexico. Note the paucity of vegetation. Columnar object in background is a ventilation shaft casing.

Figure 6.8 Close-up view of easily eroded sandy and silty mine waste from the Mesa Top Mine, Mesa Montanosa, New Mexico.

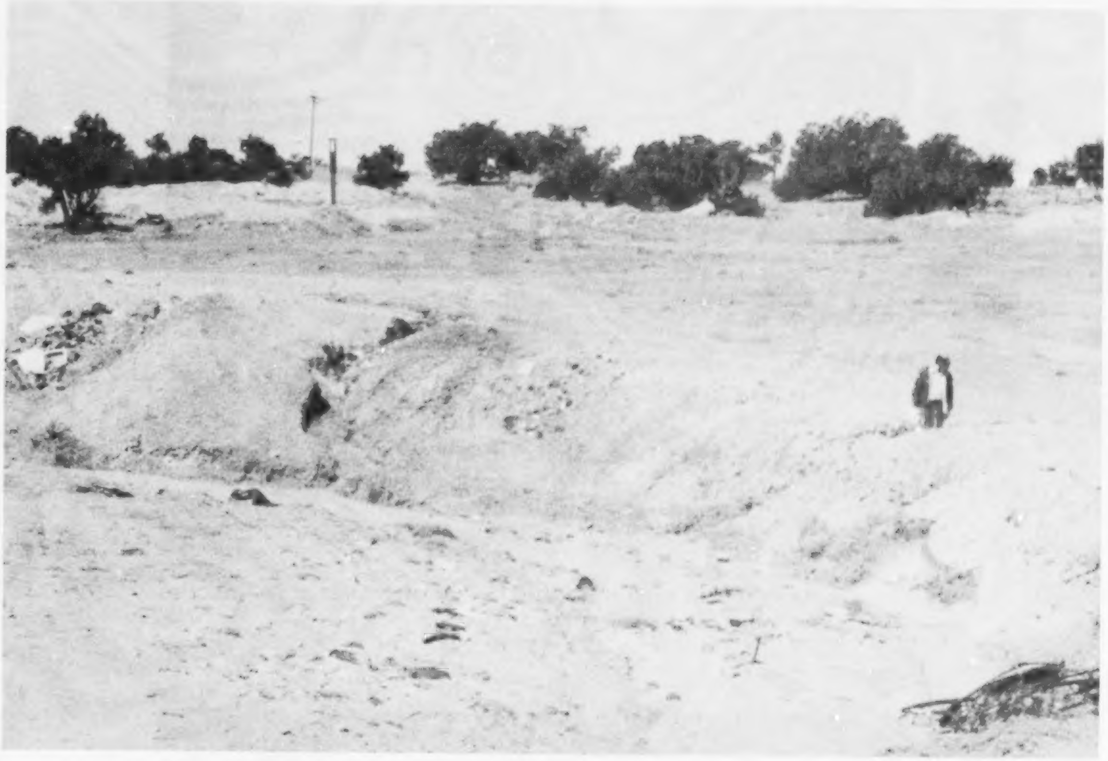

Figure 6.9 Gullying and sheet erosion of piled and spread mine wastes at the Dog Incline uranium mine, Mesa Montanosa, New Mexico.

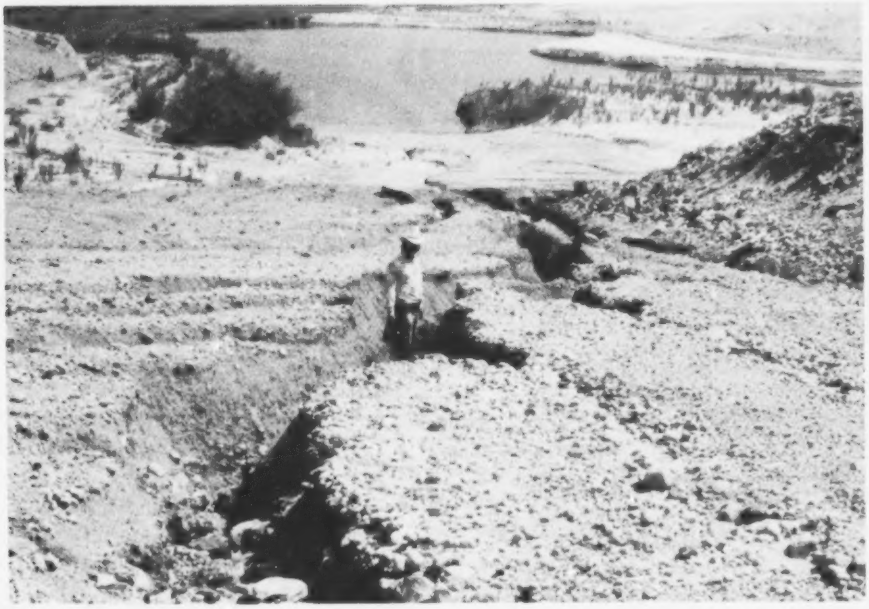

Figure 6.10 Recent erosion of unstabilized overburden piles at the inactive Galen mine, Karnes County, Texas.

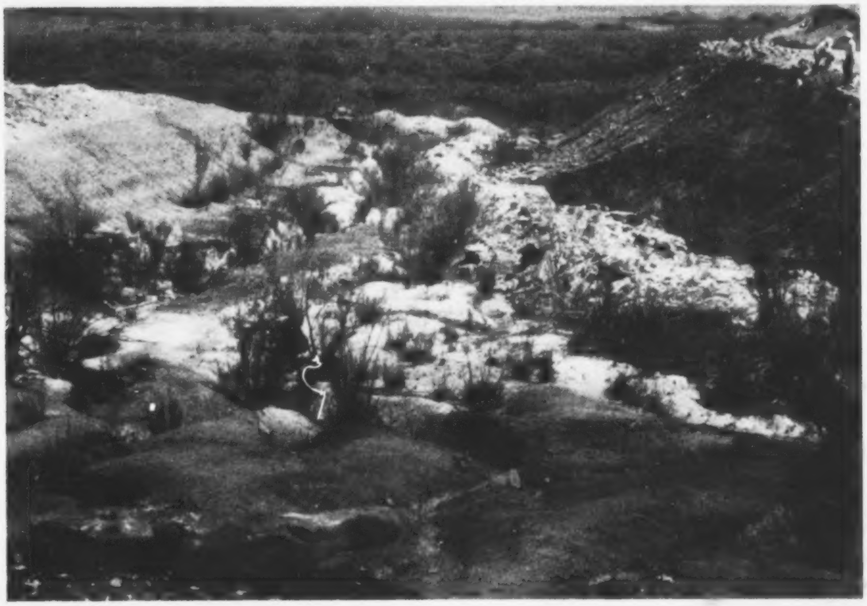

Figure 6.11 Unstabilized overburden piles and surface water erosion at the Galen Mine, Karnes County, Texas.

Mines stabilized within the last few years feature improved final contouring and use of topsoil and seeding to stimulate revegetation. The reclaimed spoil piles are then available for grazing. Because backfill cannot be complete (due to economic and bulking factors), part of the mine pit remains as shown in Figs. 6.12 and 6.13, which are of the same mine. The aerial view shows extensive patches of light colored soil devoid of vegetation. Here topsoil is missing and revegetation is minimal despite the 5 years elapsed since mining. Figure 6.13 is a closeup of one portion of the mine showing deep gullying, a thin layer of dark topsoil over relatively infertile sand and silt, and the vertical mine walls. Excavations like this must be fenced. They are a hazard to livestock and people. It is likely that erosion will continue to spread away from the mine, but the rate and consequence is unknown.

Although a mine site can be reclaimed to produce an acceptable aesthetic effect, it may not be suitable in a radiological sense. At the conclusion of surface mining, the remaining pit will contain exposed sub-ore on some of the pit walls and pit floor. Because most mines at least partly fill with water and the ore zone is thereby covered, gamma radiation and radon diffusion should be markedly reduced. Although water accumulation in the pit would be expected to have elevated concentrations of trace metals and radioactive materials, this condition would probably be temporary because of the eventual covering of the pit by sedimentation from inflow of surface water and materials sloughed from the pit walls. The natural reclamation process could be enhanced by tapering the pit walls to a more gradual slope and depositing the materials on the pit floor. If sub-ores are allowed to remain near the surface, gamma exposure rates may be sufficient to prevent unlimited land use and, even if enough stabilizing materials were used to suppress the gamma radiation, radon exhalation probably could prevent unrestricted land use also. Some of the possible radiation problems could be reduced by separating the waste rock and sub-ore when hauled to the surface. The waste rock could then be used as a blanket for the sub-ore. Away from the pit proper, surface gamma readings must be below 62 μR/hr to comply with Texas State regulations. It is reasoned that, since background is about 5 μ R/hr, surface gamma radiation of 57 μ R/hr or less would cause a total body dose of 500 mrem/yr or less.

A number of the older mines in Texas were active in the late 1950's and early 1960's--before there were requirements for stabilization. Such

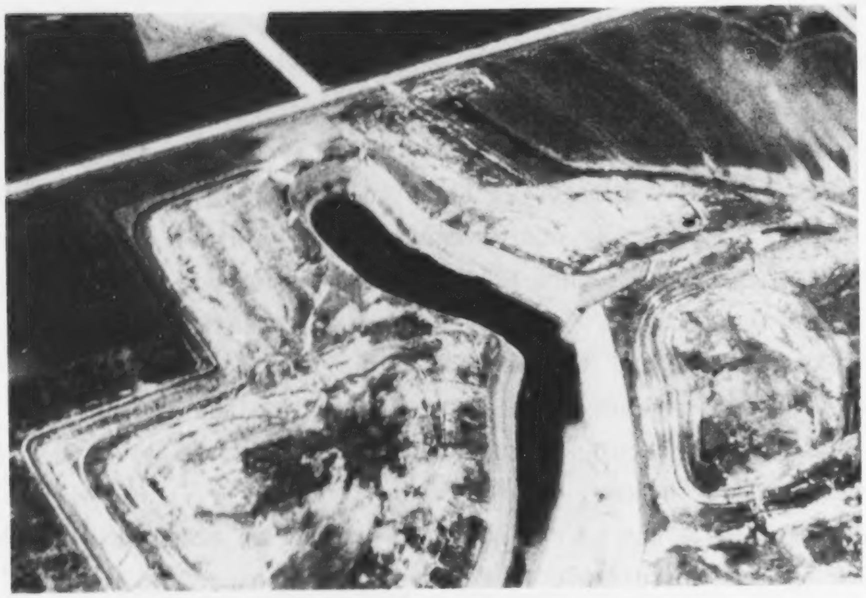

Figure 6.12 Aerial view of the Manka Mine, Karnes County, Texas. Note the extent of the mine pit and associated waste piles with poor vegetative growth on bare wastes or those with insufficient topsoil cover.

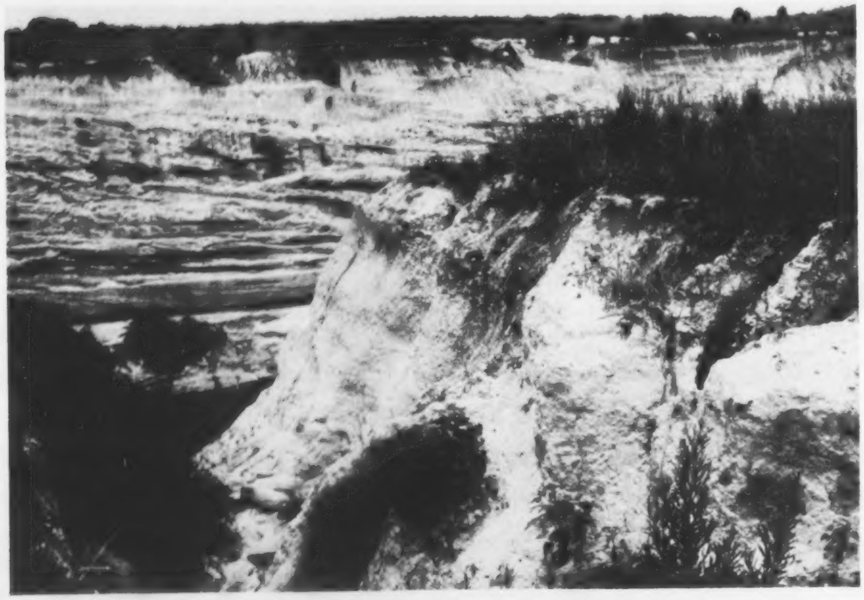

Figure 6.13 Overburden pile showing the weak vegetative cover and gullying associated with improper stabilization at the Manka Mine, Karnes County, Texas. Mine stabilized in 1974.

mines, one of which is shown in Fig. 6.14, are relatively shallow, contain shallow pools of water, and have high associated gamma radiation on the order of 80 to 100 μR/hr and as much as 140 to 250 μR/hr in some areas. The particular mine in Fig. 6.14 has maximum readings of 400 μR/hr on the mine waste piles. In addition, the mine was used for illegal disposal of toxic wastes, primarily styrene, tars, and unidentified ceramic or refraction nodules. Some of the drums containing the wastes are shown in the rear center and right of the photograph.

Mine wastes may be used for construction and other purposes if they are not controlled or restricted (see Sections 5.4 and 6.1.5.3.1). These wastes have been used for fill in a yard and park (Appendix G). Possibly they have also been used in a school area and fairgrounds (Th79). Their use in dwelling construction has also been reported (Ha74). It is also common practice to use mine wastes for road ballast and fill in areas around mine sites. This type of usage is evident from the roads immediately adjacent to and located north and northeast of the mine shown in Fig. 6.3.

In summary, only about six percent of the land used for uranium mining has been reclaimed from 1930-71 (Pa74). For the most part, the wastes at the mine sites are spreading as a result of weathering and erosion. It appears that the wastes can be controlled or disposed of by altering some mining practices, which would require very little effort or expense on the part of the mining industry. Any reclamation of the mine sites should be keyed to long-term, natural reclamation that will continue indefinitely. Careful planning can hasten the natural reclamation process and insure long-term stability of the mine sites. Measures should be taken to prevent the removal of mine wastes.

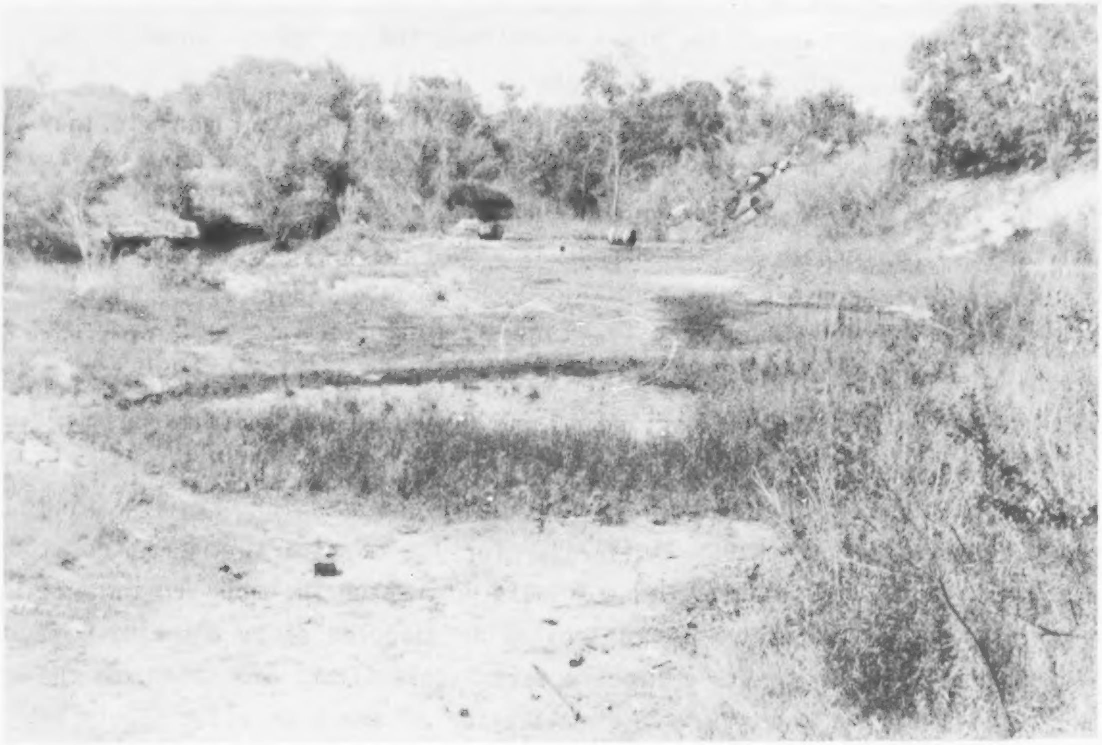

Figure 6.14 Inactive Hackney Mine, Karnes County, Texas. Drums in background contained toxic liquid wastes and styrene. Mine was active in late 1950's and early 1960.

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SECTION 7

SUMMARY AND RECOMMENDATIONS

7.0 Summary and Recommendations

7.1 Overview

This report describes the potential health and environmental effects caused by uranium mines. It considers all contaminants--solid, liquid, and airborne--and presents doses and health effects caused by wastes at both active and inactive mines. In addition to outlining the various methods of mining uranium, the report graphically depicts mine locations and lists the U.S. total of 340 active and 3,389 inactive urainum mines (Appendixes E and F) according to mine name, owner, location (state, county, section-township-range), and total ore production. Table 7.1 summarizes the mine lists.

Several facts and limitations helped shape the method and approach of this study. Little information on uranium mines is available; measurement information that is available on uranium mine wastes is frequently influenced (biased) by nearby uranium mills; there are inherent variations between uranium mines, especially between in situ mines, that complicate generic assessments of uranium mine wastes; and, finally, the law (P.L. 95-604) that mandated this study allotted only a short time in which to complete it. To accommodate these facts in our study plan, we decided to develop conceptual models of uranium mines and to make health and environmental projections from them, based upon available data from the literature; to employ conservative (maximizing) assumptions when necessary; and to supplement available information with information from discussions with persons inside and outside the agency and by doing several field studies in Texas, New Mexico, and Wyoming. Table 7.2 summarizes the sources of uranium mine contaminants that were modeled in this study.

7.2 Sources and Concentrations of Contaminants

7.2.1 Surface and Underground Mines

We calculated released radioactivity for two models of active underground and surface uranium mines. The average-large mine, the first model, reflects new and predicted future mines. The average mine, the second model, reflects the regional impact of multiple mines. The quality and

Table 7.1 Distribution of United States uranium mines by type of mine and state

State	Active				Inactive		
	Surface	Under-ground	In situ leaching	All Others[a]	Surface	Under-ground	All Other[a]
Alaska	0	0	0	0	0	1	0
Arizona	1	1	0	0	135	189	2
California	0	0	0	0	13	10	0
Colorado	5	106	0	4	263	902	52
Florida	0	0	0	0	0	0	1
Idaho	0	0	0	0	2	4	0
Minnesota	0	0	0	0	0	0	1
Montana	0	0	0	0	9	9	0
Nevada	0	0	0	0	9	12	0
New Jersey	0	0	0	0	0	1	0
New Mexico	4	35	0	3	34	142	12
N. Dakota	0	0	0	0	13	0	0
Oklahoma	0	0	0	0	3	0	0
Oregon	0	0	0	0	2	1	0
S. Dakota	0	0	0	0	111	30	0
Texas	16	0	8	1	38	0	4
Utah	13	108	0	3	378	698	17
Washington	2	0	0	0	13	0	0
Wyoming	19	6	3	2	223	32	10
Unknown	0	0	0	0	6	5	2
Total	60	256	11	13	1252	2036	101

[a] Includes mine water, heap leach dumps, miscellaneous, and unknown.

Table 7.2. Sources of contaminants at uranium mines

Source	Active Underground	Active Surface	Inactive Underground	Inactive Surface
Waste Rock (Overburden) Pile				
Wind suspended dust	M	M	M	M
Rn-222 emanation	M	M	M	M
Precipitation runoff	C	C	C	C
Sub-Ore Pile				
Wind suspended dust	M	M	M	M
Rn-222 emanation	M	M	M	M
Precipitation runoff	C	C	C	C
Ore Stockpile				
Wind suspended dust	M	M	M	M
Rn-222 emanation	M	M	M	M
Precipitation runoff	C	C	C	C
Abandoned Mine Area Surfaces				
Rn-222 emanation	M	M	M	M
Mining Activities				
Dusts	M	M	NA	NA
Combustion products	M	M	NA	NA
Rn-222	M	M	NA	NA
Wastewater				
Surface discharge	M	M	NA	NA
Seepage	C	C	C	C

Note.--M, Source modeled;C, considered but not modeled due to lack of information; NA, not applicable.

flow rates that were determined for water discharges from typical surface and underground mines in Wyoming and New Mexico, respectively, were used to calculate chemical loading of streams in three hydrographic units: subbasin (containing the mines), basin, and regional basin. Infiltration of mine water to potable groundwater and suspension/solution of contaminants in flood waters are the main components of the aqueous pathway. Crude dilution and infiltration models were used to evaluate aqueous discharge from active mines. Off-site movement from inactive mines is primarily by overland flow, the contamination significance of which was evaluated with limited field and literature surveys.

Concentrations of radionuclides and stable elements in waste rock, sub-ore, and ore, selected from only a few measurements, are shown in Table 7.3. Average annual airborne emissions for the sources listed in Table 7.2 were computed for active and inactive mines using the concentrations listed in Table 7.3 and the geological and meteorological information appropriate for each region. Source terms were maximized by assuming no dust control and no spoils pile restoration. Annual emissions of airborne contaminants estimated for the various sources are given in the following tables of Section 3.

	Tables on Active Mines		Tables on Inactive Mines	
Source	Surface	Underground	Surface	Underground
Combustion Products	3.30	3.52	--	--
Vehicular Dusts	3.32	3.56	--	--
Dust from Mining Activities	3.33	3.54	--	--
Wind Suspended Dust	3.34	3.55	3.70	3.76
Radon-222 Emissions	3.35	3.51	3.74	3.77

Annual emissions in mine water discharged to the surface by the model average underground and surface mines are listed below.

Parameter	Surface Mine (Wyoming)	Underground Mine (New Mexico)
Flow rate, m^3/min	3.0	2.0
Uranium-238, Ci/yr	0.037	0.49
Uranium-234, Ci/yr	0.037	0.49
Radium-226, Ci/yr(b)	0.00065	0.0014
Radon-222 and each short-lived daughter, Ci/yr	0.00065	0.0014
Lead-210, Ci/yr	0.00065	0.0014
Polonium-210, Ci/yr	0.00065	0.0014
Arsenic, Kg/yr	7.9	13
Barium, Kg/yr	ND(a)	850
Cadmium, Kg/yr	6.3	7
Molybdenum, Kg/yr	ND	300
Selenium, Kg/yr	ND	70
Sulfate, MT/yr(b)	276	122
Zinc, Kg/yr	112	45
Total suspended solids, MT/yr	33.0	29

(a) No data available.

(b) The values shown for radium-226 and sulfate are 10 percent and 20 percent, respectively, of those released on an annual basis. Radium is assumed to be irreversibly sorbed, and sulfate readily infiltrates.

Table 7.3. Concentration of contaminants in waste rock (overburden), ore, and sub-ore

Nonradioactive

Stable Element	Concentration, μg/g Waste Rock	Concentration, μg/g Ore and Sub-ore	Stable Element	Concentration, μg/g Waste Rock	Concentration, μg/g Ore and Sub-ore
Arsenic	9	86	Manganese	485	960
Barium	290	920	Molybdenum	2.5	115
Cadmium	NA	ND	Potassium	7,000	25,000
Cobalt	NA	16	Lead	22	78
Copper	18	61	Ruthenium	NA	ND
Chromium	< 51	20	Selenium	2	110
Iron	6,000	15,700	Strontium	150	130
Mercury	< 8	ND	Vanadium	100	1,410
Magnesium	NA	3,500	Zinc	20	29

Radioactive

Radioactive Contaminant	Concentration, pCi/g Waste rock	Concentration, pCi/g Sub-ore	Concentration, pCi/g Ore
U-238 and each daughter	6	(a)	285
Th-232 and each daughter	1	2	10

(a) The concentration of U-238 and each daughter was assumed to be 99 pCi/g at active underground mines, 40 pCi/g at active surface mines, and 110 pCi/g at inactive mines of both types.

Note.--NA, Not available; ND, Not detected.

7.2.2 In Situ Leach Mines

The sources of airborne releases that we assessed at our model in situ leach mine were the uranium recovery and packaging unit, the evaporation ponds, and the surge tank. The annual releases for these sources are listed below.

Source	Annual Airborne Release Rate
Recovery Plant	
Uranium-238	0.10 Ci
Uranium-234	0.10 Ci
Uranium-235	0.0048 Ci
Thorium-230	0.0017 Ci
Radium-226	0.00010 Ci
Lead-210	0.00010 Ci
Polonium-210	0.00010 Ci
Ammonia	3.2 MT
Ammonium chloride	12 MT
Carbon dioxide	680 MT
Surge Tank	
Radon-222	650 Ci
Storage Ponds	
Ammonia	100 MT
Ammonium chloride	300 MT
Carbon dioxide	80 MT

Since in situ mining is site specific and relatively new, little information is available on its wastes. Thus, only airborne releases were assessed quantitatively; liquid and solid wastes were discussed qualitatively.

Several chararcteristics of in situ mining, especially regarding its liquid and solid wastes, tend to minimize its release of contaminants.

First, only a small fraction of Ra-226 is leached (2.5 percent assumed); second, all liquid wastes are impounded with no planned releases; third, much of the liquid waste evaporates, except at a few sites in Texas where the wastes are injected into deep wells; and, finally, at in situ mines solid wastes accumulate at a much lower rate than they do at conventional mines. Aquifer restoration and underground excursion of the leaching solution were also discussed qualitatively. Although restoration has not yet been done at a commercial scale site, preliminary experiments indicate that proper aquifer restoration is possible. During the restoration process, Rn-222 will continue to be purged from the aquifer and should be considered a possible source of exposure.

7.2.3 Uranium Exploration

During exploration and developmental drilling, dusts are produced, Rn-222 and combustion products from drilling equipment are released, and approximately 0.2 hectares of land surface are disturbed per drill hole. The average mine site produces an estimated 6,100 kg of airborne dust, 20 kg of which is ore and subore. About 3400 Ci of Rn-222 are released annually from all development holes drilled since 1948 (4.5×10^5), which is similar to that released from one operating mine. Combustion product releases are small.

7.3 Exposure Pathways

Exposures were assessed for a hypothetical most exposed individual living about 1600 m (1-mile) from the center of the mine and for a population residing within an 80-km (50-mile) radius of the mine. The meteorological and geological parameters used were those appropriate to the respective sites.

Aqueous releases were modeled through a basin, sub-basin, and regional basin hydrographic area. Dilution by precipitation, snowmelt, and periodic flooding (typical of semiarid regions) was analyzed but not used in the model. For the model we assumed that the average annual release of contaminants is diluted by the average annual flow rate of the stream being considered. The pathways that we assessed are listed below.

Air Pathways	Water Pathways
1. Breathing a. Radioactive particulates and radon-222 b. Radon-222 daughters	1. Breathing a. Resuspended contaminants deposited from irrigation water
2. External Exposure a. Submersion b. Surface deposited radioactivity	2. External Exposure a. Submersion in resuspended contaminants deposited from irrigation water
3. Eating a. Above-surface foods grown in the area b. Milk and beef cattle grazing in the area	3. Eating a. Above-surface foods grown in the area b. Milk and beef cattle grazing in the area and drinking contaminated water c. Fish

In addition to the risks caused by wastes at or discharged directly from the mines, we assessed the risks to occupants of habitable structures built on land containing uranium mine wastes. The radium-226 in these wastes increases the concentrations of radon-222 and its decay products and the gamma radiation inside these structures.

7.4 Potential Health Effects

7.4.1 Radioactive Airborne Emissions

The risks of fatal cancer were estimated for radioactive airborne emissions. They include the lifetime risk to the most and average exposed individuals in the regional population and the number of additional fatal cancers in the regional population caused per year of model mine operation (see Table 7.4).

The major fatal cancer risk at each of the model uranium mines is the risk of lung cancer from Rn-222 daughter exposures (Tables 6.11 and 6.12). At surface and in situ mines, radioactive particulates plus Rn-222 contribute only a little over 10 percent of the total fatal cancer risk. The principal radionuclides in the airborne particulate emissions are U-238, U-234, Th-230, Ra-226, and Po-210. The contribution from Th-232 and its daughters is minor. At underground mines, essentially all the risks are due to Rn-222 daughter exposures. Fatal cancer risks at active underground mines are greater than those at active surface mines because of the larger quantity of Rn-222 daughter products released. For inactive mines, the risks are similar at surface and underground sites.

Most of the exposure to individuals around the model uranium mines is received internally, usually by breathing. However, the average person in the region around surface mines receives most of his exposure by eating contaminated foods. The largest contributors to the radioactive particulate plus Rn-222 impact are ore and overburden at active surface mines and ore and sub-ore at the active underground mines. For the model in situ mine, the uranium processing plant was the main source of particulate radionuclides.

Of all evaluated model uranium mines, the average large underground mine (Table 7.4) causes the largest fatal cancer risk and the largest number of additonal cancers in the regional population. Compared to the natural occurrence of fatal cancer from all causes (Table 7.5), we estimate an increase of 1.3 percent (0.0019) in fatal cancers over the lifetime of the maximum individual and a 0.0003 percent (0.018) increase in fatal cancers in the regional population per year. Increases in expected fatal cancers are less at all other model mine sites.

Compared to a normal occurrence of genetic effects of 0.06 effects/birth and 12.1 effects/year in the regional population (Wyoming), the computed risk of additional genetic effects from radiation exposure at the model uraninum mines is very small. The average large surface mine produces the largest increase in genetic effects. We estimate the genetic risk to the descendants of the most exposed individual to be an additional 6.4E-5 effects/birth (0.1 percent increase) for a 30-year parental exposure; 2.2E-7 effects/birth (0.00036 percent increase) to the descendants of the average exposed individual in the regional population for the same

Table 7.4 Summary of fatal cancer risks from radioactive airborne emissions of model uranium mines

Source	Most exposed individual lifetime fatal cancer risk (a)	Average exposed individual lifetime fatal cancer risk (a)	Fatal cancers cancers caused in regional population per year
Average Surface Mine	1.3E-4	2.5E-7	1.7E-4
Average Large Surface Mine	4.2E-4	8.1E-7	6.4E-4
Average Underground Mine	2.0E-4	9.1E-7	1.7E-3
Average Large Underground Mine	1.9E-3	8.6E-6	1.8E-2
Inactive Surface Mine	3.4E-5	6.3E-8	1.3E-5
Inactive Underground Mine	2.0E-5	8.6E-8	4.5E-5
In Situ Leach Mine	2.2E-4	3.9E-7	3.1E-4

(a) Lifetime exposures were calculated as follows:
Surface and underground mines: Exposure for 17 years to active mining and 54 years to inactive mine effluents.
Inactive mines: Exposure for 71 years to inactive mine effluents.
In situ leach mine: Exposure for 10-year operation and 8-year restoration.

Table 7.5 Percent additional lifetime fatal cancer risk for a lifetime exposure to the individual and the percent additional cancer deaths in the regional population per year of exposure estimated to occur as a result of uranium mining

Source	Most Exposed Individual	Average Exposed Individual	Regional Population
Average surface mine	8.7E-2	1.7E-4	7.9E-6
Average large surface mine	2.8E-1	5.4E-4	3.0E-5
Average underground mine	1.3E-1	6.1E-4	3.1E-5
Average large underground mine	1.3	5.7E-3	3.3E-4
Inactive surface mine	2.3E-2	4.2E-5	6.1E-7
Inactive underground mine	1.3E-2	5.7E-5	8.3E-7
In situ leach mine	1.5E-1	2.6E-4	1.4E-5

Note.--Comparisons are based on the risks given in Table 7.4, a national cancer risk from all causes of 0.15, and an estimate of the cancer death rate from all causes to the regional populations of New Mexico (5,400 deaths) and Wyoming (2,140 deaths).

exposure period; and 7.9E-5 additional genetic effects committed to the descendants of the regional population per year of mine operation. The latter increase is very small compared to the 12.1 effects that will normally occur each year in the live births of the regional population.

7.4.2 Nonradioactive Airborne Emissions

Atmospheric concentrations of nonradioactive air pollutants were calculated at the location of the most exposed individual. The concentrations were compared with calculated nonoccupational threshold limit values, natural background concentrations, and average urban concentrations of selected airborne pollutants in the United States.

Of the pollutant sources investigated, three produced insignificant health hazards:

1. airborne stable trace metals
2. airborne combustion products from heavy equipment operation
3. nonradioactive gas emissions at in situ leach mines

However, at active surface mines, dust particulates (produced mainly by vehicular traffic) equal or exceed conservatively calculated nonoccupational threshold limit values and, therefore, are a potential nuisance.

7.4.3 Radioactive Aqueous Emissions

The only water from active uranium mines is that pumped from the mines and released to surface streams. The largest radiation dose* from this water to individuals in the assessment regions is to the endosteal cells (bone) (see Tables 6.25 and 6.26). It primarily comes from eating foods grown on land irrigated by streams fed by discharged mine water. Significant, but of lesser importance, are exposures due to breathing wind suspended material from irrigated land, eating fish caught in streams near the site, and external gamma radiation from land irrigated by streams fed by mine water discharges. We estimate only a small risk from eating beef and milk from cattle grazing on irrigated pasture and drinking water contaminated by mine discharges ($<$ 2 percent of the total risk from aqueous emissions). The radionuclides of major importance in the risk analyses are U-238 and U-234.

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*In Section 7, "dose" is to be read as "dose equivalent"--absorbed radiation (dose) multiplied by a quality factor.

The risks of fatal cancer were estimated for radioactive aqueous discharges to surface streams from active uranium mines. The estimates included, for the 17-years of active mine operation, the cumulative risk to the most and average exposed individuals in the assessment area and the number of fatal cancers caused to persons residing within the assessment area (Table 7.6). Aqueous emissions from inactive mines and from in situ leach mines were not modeled due to a lack of data. However, we believe aqueous source terms from these mines would be low.

Drinking water may be an important source of exposure for the most exposed individual living near a uranium mine. However, we did not estimate it because we could not quantify radionuclide concentrations in potable groundwater with available information. Also, mine water probably is not consumed directly by man.

Table 7.6 Summary of the fatal cancer risks caused by radioactive aqueous emissions from model uranium mines

Source	Most exposed individual's lifetime fatal cancer risk for 17 years of mine operation	Average exposed individual's lifetime fatal cancer risk for 17 years of mine operation	Fatal cancers caused in the assessment area population from 17 years of mine operation
Underground mine site (New Mexico)	5.6E-6(3.7E-3%)[a]	3.4E-7(2.3E-4%)	2.2E-2(2.3E-4%)
Surface mine site (Wyoming)	1.2E-7(8.0E-5%)	1.6E-8(1.1E-5%)	2.6E-4(1.1E-5%)

[a] All "risks" in this table are in addition to the 0.15 risk of fatal cancer from all causes.

Although aqueous discharges from the model underground mine produce greater risks than those from the model surface mine, primarily because of greater releases of U-238 and U-238 daughters, aqueous releases at either mine cause only very small cancer risks (see Table 7.6) beyond the 0.15

natural risk of fatal cancer. For example, in New Mexico (assessment population 64,950) and Wyoming (assessment population 16,230), 9,742 and 2,434 deaths from cancer from all causes are projected to occur. Aqueous mine discharges in these areas will add only 0.022 and 0.00026 estimated deaths, respectively, to these totals.

The largest increase in estimated genetic effects occurs at the underground mine site. However, compared to the natural occurrence of hereditary disease, the overall risk of additional genetic effects due to radionuclides discharged in water from the model mines is very small. Based on a natural occurrence of 0.06 effects/birth, there will be 936 genetic effects in the regional population of New Mexico during 17 years of mine operation. In contrast, there will be only 0.015 additional effects to all the descendants of the regional population because of the 17-year exposure period.

7.4.4 Nonradioactive Aqueous Emissions

Aqueous concentrations of nonradioactive pollutants were calculated for stream water we assumed was used by the average individual within the assessment area. The pathways considered are those listed in Section 7.3. Drinking water might be a significant pathway for the most exposed individual. However, we could not make a reliable prediction of increased groundwater concentrations due to mine dewatering with the available data.

A comparison of the water concentrations of several pollutants with recommended EPA limits for livestock and irrigation usage (see Table 6.29) showed that only molybdenum from the underground mine approaches its limit for irrigation. The sums of the ratios of the average water concentrations to the recommended limits are less than one, indicating that mixtures of the metals would not exceed a "composite limit" for an average individual in the assessment areas. Constituents such as solids and sulfates, for which limits are unavailable, have minimal or no toxic properties.

More information is needed before definitive conclusions can be reached about health hazards caused by nonradioactive waterborne emissions. Uranium, the metal estimated to be in highest concentration, has no established limits based on chemical toxicity in the United States. Of particular interest would be data on water use patterns near the mines and the degree to which mine discharges may infiltrate groundwater supplies.

7.4.5 Solid Wastes

We estimated the risk of fatal lung cancer to individuals living in houses built on land contaminated by uranium mine wastes as a function of the Ra-226 concentration in the wastes (see Table 7.7). How much mine waste has been used for homesite land fill as well as its level(s) of contamination are unknown. Because of the cost, it is unlikely that mill-grade ore would be available for off-site use. It is more likely that waste rock, perhaps mixed with some sub-ore, would be the material used. Considering the Ra-226 content of sub-ores and the likelihood of its being diluted with waste rock and native soil, mine wastes in residential areas would probably contain between 5 to 20 pCi/gm of Ra-226.

Table 7.7 Estimated lifetime risk of fatal lung cancer to the average person living in a home built on land contaminated by uranium mine wastes

^{226}Ra in Soil (pCi/g)	Indoor Working Levels (WL)	Lifetime Risk of Fatal Lung Cancer[a]
5	0.02	0.025
10	0.04	0.050
20	0.08	0.10
30	0.12	0.15

[a] Based on the average individual being inside his home 75 percent of the time.

7.5 Environmental Impacts

We evaluated the environmental effects of uranium mining, including exploration, by reviewing completed studies, extensive communications with State and Federal agencies, field studies in Wyoming and New Mexico, reconnaissance visits to Wyoming, Colorado, New Mexico, and Texas, and imagery collection and interpretation. Underground and surface mines were examined to develop a sense of an average or typical condition with respect to mine size, land areas affected, quality and quantity of airborne and waterborne releases, and general, qualitative appreciation for the effects of such operations on surface streams, groundwater, disturbed land areas, and natural recovery processes. In many instances, conditions can be documented, but the significance remains highly subjective and thus weakens the justification for corrective action, particularly for inactive mines.

7.5.1 Land and Water Contamination

We conclude that (1) U.S. uranium mills make little use of mine water; (2) mine drainage is to the environment, with occasional use for agriculture, sand backfilling, construction, and potable supply; (3) active surface mines in Wyoming and underground mines in New Mexico have the greatest discharge to the offsite environment; (4) inactive surface mines do not appear to adversely affect groundwater quality, although water in such mines is typically contaminated and runoff from surface accumulation of overburden and sub-ore may be a source of surface water contamination; and (5) selected inactive underground mines in Colorado and possibly adjacent portions of Utah may discharge water enriched in radionuclides and trace elements. Since the mining industry now uses terrestrial ecosystems extensively as sinks for mining-related contaminants, an appropriate government agency should monitor active mines for groundwater quality, sorption of contaminants on stream sediments, and the flushing action of flooding events.

Before and during surface and underground uranium mining, contaminated mine water is frequently discharged to arroyos and pasture lands adjacent to the mines. Less frequently, mine water is used in nearby uranium mills, in which case ultimate disposal is to the mill tailings pile where evaporation and seepage occur. However, despite this practice of mine water discharge to land and despite the existence of over 3,000 active and inactive mines and the accelerating level of exploration and mining, there are many more studies and surveys on the interaction of uranium mills and water resources than there are on uranium mines and water resources. With few exceptions, monitoring mine water quality has been related to NPDES permits.

When mines discharge water to open lands and water courses, 90 percent or more of it infiltrates the soil and the balance evaporates. Stable and radioactive contaminants subject to sorption are selectively concentrated in nearby soils, which become a local sink. Mobile constituents such as sulfate and chloride probably percolate to the water table along with the bulk of the water, which recharges nearby shallow aquifers downgrade from the mines. Although many areas in New Mexico, Texas, Colorado, Wyoming, and Utah have received mine water discharge, studies of contaminant accretion on soils and deterioration of groundwater quality have been rather limited. Widespread contamination of groundwater has not been documented, but there are indications that local surface water and groundwater quality have been adversely effected in Colorado, Wyoming, and Texas. Studies underway in New Mexico reveal, in at least two mining districts, groundwater deteriorating because of mine drainage. Significant increases in ambient uranium and radium occurred in the Shirley Basin uranium district of Wyoming because of initial strip mining and mill processing and, to a lesser extent, in situ leaching. The long-term significance of soil loading with stable and radioactive contaminants and their cycling through the terrestrial ecosystem, including the human food chain, has not been determined for uranium mining operations.

Discharges from model active surface and underground mines average 2 to 3 m^3/minute. In most cases, complete infiltration takes place in stream beds within 5 to 10 kilometers of the mines. However, when discharges from several mines are combined or if single mine discharge is several cubic meters per minute or more, infiltration and storage capacity of the alluvium in nearby channels is exceeded and perennial flows are created for distances of 20 to 30 kilometers. For example, underground uranium mines in the Grants Mineral Belt of New Mexico currently discharge 66 m^3 per minute. Of this, only 12 m^3 per minute are used in uranium mills; the balance is discharged to nearby washes or arroyos. Fourteen of the 20 active uranium mills make no use of mine water, which is associated with essentially every active underground mine and most active surface mines, particularly in Texas and Wyoming.

Annual contaminant loading from continuous discharge at a rate of 3 m^3/ minute from one surface mine in the Wyoming model area and dilution in flood flows with recurrence intervals of 2 to 25 years produce the loading and stream concentration values in Table 7.8. Chemical loading was calculated on a mass-per-time basis to estimate the effects of mine drainage.

For assessing environmental impacts, we assume that most contaminants remain on or near the land surface and are available for resuspension in periodic flash flooding in the sub-basin. Sorption, precipitation, and so on are assumed to render 90 percent of the radium-226 unavailable for further transport. Eighty percent of the sulfate is assumed to infiltrate and also becomes unavailable for further transport in flood waters.

Stream concentrations for uranium, zinc, cadmium, and arsenic are likely to be less than those shown because there will not be 100 percent resuspension of sorbed contaminants, and flood events with lesser return periods are also likely to disperse contaminants. The loading data are believed to be quite realistic; it is the temporal distribution and re-distribution of the contaminants that constitute a significant unknown. These preliminary results indicate contamination of surface water with uranium, radium, sulfate, and, to a lesser extent, with cadmium and arsenic in stream waters near the mine outfall. Subsequent dilution of these initial concentrations will occur as the flow merges with that of progressively larger streams in the downgrade direction, but cadmium and sulfate may exceed the drinking water standard in flood waters as far as the regional basin. Impoundment of these initial flows can be expected considering water management practices in semiarid rangeland areas like Wyoming. Therefore, further pathway investigations, based on field data, are needed.

For the model underground mining area, we selected the Ambrosia Lake District of New Mexico. We assumed that 14 mines discharged an average of 2 m^3/minute and that loading took place for two years prior to each flood. We then calculated concentrations in flood water for eight different cases-for 2, 5, 10, and 25 year floods (larger numbers indicating larger floods), with concentrations for each flood being calculated on the basis of both a 1-day and 7-day flood duration (see Table 7.9). Based upon these assumptions and calculations, it appears that concentrations in flood waters, particularly in the basin, may exceed established or suggested standards for uranium, radium, cadmium, arsenic, selenium, barium, and sulfate. However, precipitation and sorption, in addition to dilution farther downstream, probably will reduce these concentrations enough so that quality standards for drinking and irrigation water can be met. But

Table 7.8 Summary of contaminant loading and stream water quality from a model surface uranium mine

Annual Loading Per Mine [a] (Kg/yr)	Drinking Water Standard (mg/ℓ)	Concentrations in Basin and Regional Basin Flood Flows for Floods of 2, 25, and 100 Years Return Period, mg/ℓ			
		Basin		Regional Basin	
		Min	Max	Min	Max
Uranium 110	0.015/3.5/0.21[b]	0.36	0.76	0.26	0.44
Radium-226 0.00065 Ci/yr	5 pCi/ℓ	2.1 pCi/ℓ	4.5 pCi/ℓ	1.6 pCi/ℓ	2.6 pCi/ℓ
TSS 32,955	---	107	228	79	131
Sulfate 2.76×10^5	250	900	1909	668	1098
Zinc 112.0	5.0	0.366	0.774	0.271	0.445
Cadmium 6.31	0.01	0.02	0.044	0.015	0.025
Arsenic 7.88	0.05	0.025	0.054	0.019	0.031

[a] Loading values shown for radium and sulfate are reduced to 10 percent and 20 percent, respectively, of the amount actually released by a mine. Irreversible sorption and precipitation affect radium and sulfate infiltrates to the water table.

[b] 0.015 mg/ℓ : Suggested maximum daily limit based on radiotoxicity for potable water consumed at a rate of 2 liters per day on a continuous basis. 3.5 mg/ℓ : Suggested maximum daily limit based on chemical toxicity and intake of 2 liters in any one day. 0.21 mg/ℓ : Suggested maximum daily limit based on chemical toxicity and intake of 2 liters per day for 7 days.

Table 7.9 Summary of contaminant loading and stream water quality from a model underground uranium mine

Annual Loading, Per Mine(a) (Kg/yr)	Drinking Water Standard (mg/ℓ)	Concentrations in Basin and Regional Basin for 1-day and 7-day Floods of 2 to 25 Years Return Period, mg/ℓ			
		Basin		Regional Basin	
		Min	Max	Min	Max
Uranium 1480	0.015/3.5/0.21(b)	6.9	7.1	0.045	0.046
Radium-226					
0.0014 Ci/yr	5 pCi/ℓ	6.7 pCi/ℓ	6.9 pCi/ℓ	0.044 pCi/ℓ	0.044 pCi/ℓ
Lead-210					
0.0014 Ci/yr	---	71.2 pCi/ℓ	73.4 pCi/ℓ	0.470 pCi/ℓ	0.0472 pCi/ℓ
Cadmium 7	0.01	0.03	0.03	0.0002	0.0002
Arsenic 13	0.05	0.061	0.063	0.00039	0.00041
Selenium 80	0.01	0.37	0.38	0.0026	0.0026
Molybdenum 300	---	1.4	1.4	0.0089	0.0093
Barium 850	1.0	4.0	4.2	0.26	0.27
Zinc 45	5.0	0.21	0.22	0.0014	0.0014
Sulfate 1.22×10^5	250	574	584	3.7	3.8
TSS 29,000	---	130	140	0.89	0.92

(a)Loading values shown for radium and sulfate are reduced to 10 percent and 20 percent, respectively, of the amount actually released by a mine. Irreversible sorption and precipitation affect radium and sulfate infiltrates to the water table.

(b)0.015 mg/ℓ : Suggested maximum daily limit based on radiotoxicity for potable water consumed at a rate of 2 liters per day on a continuous basis. 3.5 mg/ℓ : Suggested maximum daily limit based on chemical toxicity and intake of 2 liters in any one day. 0.21 mg/ℓ : Suggested maximum daily limit based on chemical toxicity and intake of 2 liters per day for 7 days.

more theoretical and field evaluations are needed to confirm this.

In situ leaching has contaminated local groundwater reservoirs. We expect that this will continue because leach solution excursions from the well field do occur and because injected constituents, especially ammonium, can not be fully recovered. The NRC and agreement States recognize this situation but consider the adverse impacts outweighed by the benefits of recovering additional uranium and developing a relatively new technology.

7.5.2 Effects of Mine Dewatering

Underground mines and most surface mines are dewatered to allow for excavation or shaft sinking and ore removal. The resulting low concentration and, oftentimes, large volume effluent discharges introduce substantial masses of stable and radioactive trace elements to local soil and water systems. This extensive use of soils in both the saturated and unsaturated zones as water and contaminant sinks requires further study to determine the environmental fate of those elements. In addition to local effects, the long-term impacts on regional water availability and quality are also important. The NPDES limits relating to surface discharges are, in terms of parameters and concentrations, different from one EPA region to another and should be reevaluated to more closely reflect the impact of contaminant concentration and mine discharge. In general, the uncertainties about the environmental impact of mine dewatering can be expected to increase; and additional, comprehensive investigations of its effects are necessary.

7.5.3 Erosion of Mined Lands and Associated Wastes

From initial exploration through retirement, mining, particularly surface mining, increases erosion and sediment yield. The most significant waste sources are access roads, drilling pads, and piles of overburden/waste rock and sub-ore. Sediment and associated contaminants are dispersed mostly through the overland flow of precipitation and snowmelt water. Erosion rates vary considerably with the characteristics of the source area, i.e., pile geometry, soil and rock characteristics, amount and type of vegetative cover, topography, and local climate. There is some erosion of all mine waste sources, although studies of ephemeral drainage courses downgrade from inactive mines in New Mexico and Wyoming usually reveal only local soil and water contamination and no significant off-site dispersal of contaminants. Proper reclamation, particularly grading and revegetation, markedly reduce erosion and, consequently, contaminant transport.

7.5.4 Exploratory and Development Drilling

The uranium industry has drilled approximately 1,300,000 exploratory and development drill holes through 1977. This amounts to about 430 drill holes per mine if averaged over all active and inactive mines. During the course of drilling, some land areas are disturbed to provide access roads to the drill sites and pads for the drill-rig placements. This has disturbed about 2500 km^2 (960 mi^2) of land for all drilling through 1977.

Drilling wastes accumulate at each drill site. Although these wastes are sometimes placed in trenches and backfilled after drilling, the general industry practice (observed from field studies and aerial photography), apparently, is to allow the wastes to remain on the surface, subject to erosion. The extent of radiological contamination from erosion of the remaining ore and sub-ore at development drill holes is not known.

The average drilling depth has increased with time and will probably continue to do so in the future. Deeper drilling will tend to increase the probability that several aquifers may be penetrated by each drill hole. Aquifers with good quality water may be degraded by being connected, via the drill holes, with aquifers of poor quality water. Current regulations require drill holes to be plugged to prevent interaquifer exchange, but often only the first one and one-half meters of the borehole will be plugged, and regulations do not effect past drill holes. Finally, it appears, from mine site surveys and aerial photography, that very few drill sites have been reclaimed.

7.5.5 Underground Mining

The land disturbed by individual underground mines varies from 0.89 to 17 hectares (2.2 to 42 acres) with an average of 9.3 hectares (23 acres). In addition, access roads to the mines consume about 1.1 hectares (2.7 acres), and mine subsidence disturbs about 1.5 hectares (3.7 acres). A total of about 12 hectares (30 acres) of land are disturbed by an average underground mine.

All underground uranium mining through 1977 has produced about 2.9 x 10^7 MT or about 1.8 x 10^7 m^3 of wastes. Some of these wastes, the sub-ores, contain elevated concentrations of naturally occurring radionuclides. The sub-ores usually are removed last in the mining process and dumped on top of the waste rock where they are subject to erosion. Some radiation surveys conducted around waste piles indicate that the sub-ores are eroding and contaminating land in addition to that disturbed by the mining activities.

During our field studies in Texas, New Mexico, Wyoming, and Colorado, we saw very few mine sites where reclamation had been completed or was in progress--especially at the inactive mine sites.

7.5.6 Surface Mining

The cumulative waste from surface mining uranium between 1950 and 1978 amounts to about 1.7 x 10^9 MT (1.1 x 10^9 m^3). Overburden is usually used to backfill mined-out pits during contemporary mining. At older inactive mines, the mine wastes were either used for pit backfill or completely disregarded. Erosion of these waste piles may cause substantial environmental problems.

The amount of land physically disturbed at a surface mine is highly variable. The area disturbed at ten surface mines was estimated to range from 1.1 to 154 hectares (2.7 to 380 acres), averaging about 41 hectares (101 acres) per mine site. Access roads disturb about 3 hectares (7.4 acres) per mine site, bringing the total average area physically disturbed to about 44 hectares (109 acres). Field surveys of inactive mine sites indicate that mine wastes (sub-ores) erode and contaminate land areas greater than those physically disturbed. The land contamination appears to have been caused by erosion of ore stockpiles, erosion of sub-ores, and dust losses from the actual mining process.

Very few if any inactive mine sites were reclaimed. Reclamation of any mine site will have to address the radiological aspects of the mine and its wastes.

7.6 Regulatory Perspective

Except for in situ leach mining, licensed by the Nuclear Regulatory Commission (NRC), uranium mining is not licensed, per se, by a Federal agency. However, three Federal statutes have particular relevance to uranium mining. First, the Federal Water Pollution Control Act as amended (1972) requires a permit for discharges to navigable waters. Second, the Clean Air Act amendments of 1977 require a permit for pollutant air emissions. Third, proposed regulations under the Resource Conservation Recovery Act of 1977 identify hazardous wastes and stipulate their disposal for uranium mining. When promulgated, these latter regulations will strengthen current Federal and State reclamation requirements.

In situ uranium mining is licensed by those states having agreement-state status with NRC. National Pollution Discharge Elimination System (NPDES) permits are issued by EPA approved states. No state issues mining licenses per se. However, most states require mining and reclamation plans, including bonding fees, for at least state-controlled lands. Most reclamation requirements provide erosion control through slope and vegetation standards. Arizona is the only uranium mining state without reclamation requirements.

7.7 Conclusions and Recommendations

The evaluation of the potential impacts of uranium mining was performed largely by means of analytical studies of model facilities. We believe that the results give an adequate representation of the industry. In order to determine the extent of possible problems, our studies were specifically designed to give conservative results. It should be recognized that actual mines may operate under conditions producing substantially smaller impacts than the results presented.

Compared to uranium milling, health and environmental effects of mining are not as well understood, despite the existence of over 3000 active and inactive mines. We have noted throughout this report instances of the absence or inadequacy of pertinent information.

7.7.1 Conclusions

7.7.1.1 Solid Wastes

Solid uranium mining wastes are potentially hazardous when used as building materials or when buildings are constructed on land containing such wastes. The hazard arises principally from increased risk of lung cancer due to radon-222. In a 1972 survey of communities in uranium mining regions, EPA and the former Atomic Energy Commission found more than 500 locations where such wastes had been used.

7.7.1.2 Airborne Effluents

a) Individuals living very near active underground mine exhaust vents would have an increased risk of lung cancer caused by exposure to radon-222 emissions. Surface mines and in situ mines are less hazardous, and inactive mines do not have significant radon-222 emissions. Other airborne radioactive emissions from all types of mines are judged to be smaller.

b) The number of additional cancers committed per year in the regional populations due to radionuclide air emission from the approximately 340 active mines and 3300 inactive mines was estimated to be about 0.6 cancers in 1978. This number of estimated additional cancers is small, about one-third of the estimated additional cancers in regional populations due to radon emissions from the 24 inactive uranium mill tailings piles addressed by Title I of the Uranium Mill Tailings Radiation Control Act (EPA 80). (These mill tailings piles represent about 13 percent of all tailings currently existing due to U.S. uranium milling and mining). These potential effects are not of sufficient magnitude to warrant corrective measures, especially considering the large number of sites involved.

c) The following were judged to cause an insignificant health risk for all types of mines:

1. airborne nonradioactive trace metals.
2. airborne combustion products from heavy-duty equipment operations.
3. nonradioactive emissions from in situ leach sites.

d) Airborne dust near large surface mines (primarily caused by vehicular traffic) may exceed the National Ambient Air Quality Standard for particulate matter.

7.7.1.3 Waterborne Effluents

a) We estimate that an insignificant health risk accrues to populations from waterborne radioactivity from an average existing mine.

b) Uranium mine dewatering and water discharges, which are increasing as more and deeper mines are created, may in the future have significant effects on water quality. Current treatment practices are controlling the release of radioactivity into surface waters.

c) Water in inactive surface and underground mines usually contains radionuclides and trace elements in concentrations comparable to groundwater in contact with ore bodies. Some abandoned underground mines in certain areas of Colorado and Utah probably discharge such waters to nearby streams and shallow aquifers. Available data is not sufficient to conclude whether or not there is a problem.

d) We could not determine, using models, that there is no health hazard to individuals who drink water drawn from such surface or underground sources. Water discharges from active mines to nearby streams and stream channels may extensively recharge shallow aquifers, many of which are either now used or could be used for drinking water. Such determinations must be made on a site-specific basis, and take account of the additive effects of multiple mines. These studies can be made easily a part of State or utility surveillance programs.

7.7.1.4 Exploratory and Development Drilling

Harm from effluents due to exploratory and developmental drilling is probably small compared to effects of operating mines. Under current regulations and practices, however, aquifers penetrated at different levels can mix, creating the potential for degrading high quality groundwater.

7.7.2 Recommendations to Congress

1) Based on this study, we do not believe at this time that Congress needs to enact a remedial action program like that for uranium mill tailings. This is principally because uranium mine wastes are lower in radioactivity and not as desirable for construction purposes as uranium mill tailings. Nonetheless, some mining waste materials appear to have been moved from the mining sites but not to the extent that mill tailings were.

2) Some potential problems were found that might require regulatory action but none of these appear to require new Congressional action at this time.

7.8 Other Findings

1) Regulations may be needed to control wastes at active uranium mines to preclude off-site use and to minimize the health risks from these materials. These regulations would need to address the use of the materials for construction purposes as well as ultimate disposal of the materials.

EPA proposed such regulations in 1978 under the Resource Conservation and Recovery Act (RCRA). In 1980, Congress amended RCRA to require further EPA studies before promulgating general regulations for mining wastes. An EPA study by the Office of Solid Wastes on all types of mines, including uranium mines, is currently being conducted. The amendment did not restrict EPA's authority to regulate use of uranium mine wastes in construction or reclamation of lands containing such wastes.

2) Standards are needed to control human exposure from radioactive air emissions from uranium mines. This is principally because of potential exposure to individuals living near large underground uranium mines rather than concerns regarding the exposure of regional populations. We have proposed such standards under Section 112 of the Clean Air Act.

3) EPA has conducted two field studies in 1972 and 1978 which define possible sites at which mine wastes may have been used in construction or around buildings. The information developed in these studies has been sent to State health departments. The States should conduct follow-up studies, as appropriate, to determine whether there are problems at these sites.

4) The adequacy with which NPDES permits protect individuals who may obtain drinking water near the discharge points for uranium mine dewatering should be evaluated by States. Under the Public Water Systems provision of the Safe Drinking Water Act, radionuclide standards now exist for drinking water.

5) Some site specific studies should be considered by States to determine the extent to which inactive uranium mines are significant water pollution sources.

6) States with uranium mines should determine the feasibility of control of fugitive dust from large surface mines and incorporate the recommendations in State Implementation Plans.

7) States should require borehole plugs in drilling operations that will prevent interaquifer mixing (exchange) and also seal drilling holes at the surface.

7.9 References

EPA80 U.S. Environmental Protection Agency, 1980, "Draft Environmental Impact Statement for Remedial Action Standards for Inactive Uranium Processing Sites (40 CFR 192), "EPA 520/4-80-011.